

An aerial photograph of an oil spill in the ocean. The spill is visible as a large, irregular, light-colored slick that stretches across the dark water. The slick has several distinct, swirling patterns, suggesting it is being moved by currents or wind. The overall image is in black and white, with high contrast between the dark water and the light-colored oil.

Exxon Valdez
Oil Spill
Symposium

February 2-5, 1993
Anchorage, AK, USA

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CONTENTS

Plenary Session

So Why Can't Science Tell Us More About the Effects of the Exxon Valdez Oil Spill?

Robert B. Spies, *Applied Marine Sciences* 1

Fate of the Oil Spilled From the T/V Exxon Valdez in Prince William Sound, Alaska

Douglas A. Wolfe², M.J. Hameedi¹, J.A. Galt¹, G. Watabayashi¹, Jeffrey W. Short¹,
C.E. O'Clair, Stanley Rice¹, J. Michel², James R. Payne³, Joan F. Braddock⁴,
S. Hanna⁵, and David M. Sale⁶, ¹National Oceanic and Atmospheric Administration;
²Research Planning, Inc.; ³Sound Environmental Services Inc.; ⁴University of Alaska Fairbanks;
⁵Sigma Research Corporation; ⁶SnowOtter Environmental Consulting 6

The Exxon Valdez Cleanup: The First Six Months

Vice Admiral Clyde E. Robbins, *U.S. Coast Guard* 10

Cultural Resources and the Exxon Valdez Oil Spill

Judith E. Bittner, *Alaska Department of Natural Resources* 13

Subsistence

James A. Fall, *Alaska Department of Fish and Game* 16

Overview of Intertidal Processes, Damages, and Recovery

Charles H. Peterson, *University of North Carolina* 19

Subtidal Oil Contamination and Biological Impacts

Stanley Rice, Bruce A. Wright, Jeffrey W. Short, and C.E. O'Clair
National Oceanic and Atmospheric Administration 23

Summary of Injury to Fish and Shellfish Associated With the Exxon Valdez Oil Spill

Charles P. Meacham and Joseph R. Sullivan,
Alaska Department of Fish and Game 27

How Do You Fix the Loss of Half a Million Birds?

D. Michael Fry, *University of California Davis* 30

Effects of the Exxon Valdez Oil Spill on Marine Mammals in Prince William Sound

Kathryn J. Frost¹, Brenda E. Ballachey², Marilyn E. Dahlheim³,
and Thomas R. Loghlin³
¹Alaska Department of Fish and Game; ²U.S. Fish and Wildlife Service;
³National Oceanic and Atmospheric Administration 34

Restoration Planning Following the Exxon Valdez Oil Spill

John A. Strand, *National Oceanic and Atmospheric Administration* 37

Fate and Toxicity

Tracking Exxon Valdez Oil from Beach to Deep-water Sediments of Prince William Sound, Alaska

Paul R. Carlson and Keith A. Kvenvolden, *U.S. Geological Survey* 43

Characterization of Residual Oil in Prince William Sound, Alaska—3.5 Years Later <i>Paulene O. Roberts¹, Charles B. Henry Jr.¹, Edward B. Overton¹, and Jacqueline Michel²</i> ¹ Louisiana State University; ² Research Planning, Inc.	46
Toxicity of Intertidal and Subtidal Sediments Contaminated by the Exxon Valdez Oil Spill <i>Douglas A. Wolfe¹, Margaret M. Krahn¹, Ed Casillas¹, K. John Scott², John R. Clayton, Jr.², John Lunz², James R. Payne³, and Timothy A. Thompson²</i> ¹ National Oceanic and Atmospheric Administration; ² Science Applications International Corp.; ³ Sound Environmental Services, Inc.	48
Microbial Activity in Sediments Following the T/V Exxon Valdez Oil Spill <i>Joan F. Braddock¹, Jon E. Lindstrom¹, Thomas R. Yeager¹, Brian T. Rasley¹, Gregory Winter², and Edward J. Brown³</i> ¹ University of Alaska Fairbanks; ² Alaska Department of Environmental Conservation ³ ; University of Northern Iowa	52
Contamination of Subtidal Sediments by Oil From the Exxon Valdez in Prince William Sound, Alaska <i>C.E. O'Clair, Jeffrey W. Short, and Stanley Rice</i> <i>National Oceanic and Atmospheric Administration</i>	55
Determination of Petroleum-derived Hydrocarbons in Seawater Following the Exxon Valdez Oil Spill I: Analysis of Seawater Extracts <i>Jeffrey W. Short and Patricia Rounds</i> <i>National Oceanic and Atmospheric Administration</i>	57
Methods for Determining Crude Oil Contamination in Sediments and Biota After the Exxon Valdez Oil Spill <i>Margaret M. Krahn, Gina M. Ylitalo, Douglas G. Burrows, Jon Buzitis, Sin-Lam Chan, and Usha Varanasi</i> <i>National Oceanic and Atmospheric Administration</i>	60
Qualitative and Quantitative Determination of Exxon Valdez Crude Oil in Sediment Samples Using Principal Component Analysis of Hydrocarbon Data <i>Jeffrey W. Short and Ronald A. Heintz, National Oceanic and Atmospheric Administration</i>	63
Nearshore Subtidal Transport of Hydrocarbons and Sediments Following the Exxon Valdez Oil Spill <i>David M. Sale¹, James Gibeau², and Jeffrey W. Short³</i> ¹ SnowOtter Environmental Consulting; ² University of Texas at Austin; ³ National Oceanic and Atmospheric Administration	64
Clean-up and Treatment Effects Short-term Biological Effects of Shoreline Treatment on Intertidal Biota Exposed to the Exxon Valdez Oil Spill <i>D.C. Lees¹, W.B. Driskell², and J.P. Houghton³</i> ¹ Ogden Environmental and Energy Services Company; ² Consultant; ³ Pentec Environmental, Inc.	69
Growth and Survival of the Predatory Snail <i>Nucella lamellosa</i> in Areas Exposed to the Exxon Valdez Oil Spill <i>T. Ebert¹, D.C. Lees², and H. Cumberland²</i>	

¹San Diego State University; ²Ogden Environmental and Energy Services Co. Inc. 73

**Recovery of Prince William Sound Intertidal Infauna from
Exxon Valdez Spill and Treatments 1990-1992**

J.P. Houghton¹, A.K. Fukuyama¹, W.B. Driskell², D.C. Lees³,
Gary Shigenaka⁴, and A.J. Mearns⁴

¹Pentec Environmental, Inc.; ²Consultant; ³Ogden Environmental and Energy
Services Company, Inc.; ⁴National Oceanic and Atmospheric Administration 75

**Recovery of Prince William Sound Intertidal Epibiota
from the Exxon Valdez Spill and Treatments—1990-1992**

J.P. Houghton¹, A.K. Fukuyama¹, D.C. Lees², W.B. Driskell², A.J. Mearns⁴,
and Gary Shigenaka⁴, ¹Pentec Environmental, Inc.; ²Ogden Environmental
and Energy Services Company, Inc.; ³Consultant; ⁴National Oceanic
and Atmospheric Administration 79

**NOAA's Long-term Ecological Recovery Monitoring Program:
Overview and Implications of Recovery Trends and Treatment Effects**

Alan J. Mearns and Gary Shigenaka, National Oceanic and Atmospheric Administration 83

Subtidal

**The Effects of the Exxon Valdez Oil Spill on Benthic Invertebrates in
Silled Fjords in Prince William Sound**

Stephen C. Jewett¹, Thomas A. Dean², and David R. Laur³

¹University of Alaska Fairbanks; ²Coastal Resources Associates, Inc.;

³University of California Santa Barbara

..... 87

**The Effects of the Exxon Valdez Oil Spill on Epibenthic Invertebrates
in the Shallow Subtidal**

Thomas A. Dean¹ and Stephen C. Jewett²

¹Coastal Resources Associates, Inc.; ²University of Alaska Fairbanks 91

The Effects of the Exxon Valdez Oil Spill on Eelgrass and Subtidal Algae

Thomas A. Dean¹, Michael S. Stekol², and Stephen C. Jewett³

¹Coastal Resources Associates, Inc.; ²University of Alaska Southeast;

³University of Alaska Fairbanks 94

**The Effects of the Exxon Valdez Oil Spill on Infaunal Invertebrates
in the Eelgrass Habitat of Prince William Sound**

Stephen C. Jewett¹ and Thomas A. Dean²

¹University of Alaska Fairbanks; ²Coastal Resources Associates, Inc. 97

Salmon

Assessment of Injury to Pink Salmon Eggs and Fry

Brian G. Bue, Samuel Sharr, Steve D. Moffitt, and Andrew Craig

Alaska Department of Fish and Game 101

**Cytochrome P450 Induction and Histopathology in Pre-emergent
Pink Salmon from Oiled Streams in Prince William Sound, Alaska**

Michael Wiedmer¹, Mark J. Fink¹, and John J. Stegeman²

¹Alaska Department of Fish and Game; ²Woods Hole Oceanographic Institution 104

**Application of Half Length Coded-wire Tags to Wild and Hatchery
Pink Salmon Fry in Prince William Sound, 1989-1991**

Daniel Sharp, Carol Peckham, and Jodi Smith

Alaska Department of Fish and Game 108

Impacts of the Exxon Valdez Oil Spill on the Migration, Growth, and Survival of Juvenile Pink Salmon in Prince William Sound <i>Mark Willette, Alaska Department of Fish and Game</i>	112
The Impact of the Exxon Valdez Oil Spill on Juvenile Pink and Chum Salmon and Their Prey in Nearshore Marine Habitats <i>A.C. Wertheimer, A.G. Celewycz, M.G. Carls, and M.V. Sturdevant</i> <i>National Oceanic and Atmospheric Administration</i>	115
Effect of Oil-contaminated Food on the Growth of Juvenile Pink Salmon, <i>Oncorhynchus gorbuscha</i> <i>M.G. Carls¹, L. Holland, M. Larsen¹, J.L. Lum¹, Dorothy Mortenson¹, S.Y. Wang², and A.C. Wertheimer¹, ¹National Oceanic and Atmospheric Administration; ²University of Southern Mississippi</i>	118
Coded Wire Tag Recovery Results for Prince William Sound Pink Salmon, 1989-1991 <i>Carol Peckham, Samuel Sharr, and David G. Evans</i> <i>Alaska Department of Fish and Game</i>	121
Pink Salmon Spawning Escapement Estimation in Prince William Sound <i>Samuel Sharr, Daniel Sharp, and Brian G. Bue</i> <i>Alaska Department of Fish and Game</i>	124
A Life History Approach to Estimating the Damage to Prince William Sound Pink Salmon from the Exxon Valdez Oil Spill <i>Harold J. Geiger¹, Brian G. Bue¹, Sam Sharr¹, A.C. Wertheimer², and T. Mark Willette¹</i> <i>¹Alaska Department of Fish and Game; ²National Oceanic and Atmospheric Administration</i>	128
Overescapement Impacts of Kenai River Sockeye Salmon <i>Dana C. Schmidt, Ken Tarbox, Gary Kyle, Bruce King, Linda Brannian, and Jeff Koenings, Alaska Department of Fish and Game</i>	132
Birds	
Effects of the T/V Exxon Valdez Oil Spill on Murres: A Perspective from Observations at Breeding Colonies <i>D.R. Nysewander, C. Dipple, G.V. Byrd, and E.P. Knudtson</i> <i>U.S. Fish and Wildlife Service</i>	135
How Long to Recovery for Murre Populations, and Will Some Colonies Fail to Make the Comeback? <i>Dennis Heinemann, Manomet Bird Observatory, Inc.</i>	139
Effects of the Exxon Valdez Oil Spill on Bald Eagles <i>Timothy D. Bowman and Philip F. Schempf, U.S. Fish and Wildlife Service</i>	142
Effects of the Exxon Valdez Oil Spill on Pigeon Guillemots (<i>Cepphus columba</i>) in Prince William Sound, Alaska <i>K.L. Oakley and K.J. Kuletz, U.S. Fish and Wildlife Service</i>	144
Quantification of Habitats in Prince William Sound from Landsat Thematic Mapper Satellite Imagery <i>Richard Podolsky, GALA Software</i>	147
Effects of the Exxon Valdez Oil Spill on Marbled Murrelets <i>K.J. Kuletz, U.S. Fish and Wildlife Service</i>	148

Acute and Sublethal Effects of the Exxon Valdez Oil Spill on Harlequins and Other Seaducks

Samuel M. Patten, Jr., Alaska Department of Fish and Game 151

Black Oystercatchers in Prince William Sound: Oil Spill Effects on Reproduction and Behavior

Brian E. Sharp¹ and Mary Cody², ¹Ecological Perspectives;

²U.S. Fish and Wildlife Service 155

Effects of the Exxon Valdez Oil Spill on Black-legged Kittiwakes in Prince William Sound

David B. Irons, U.S. Fish and Wildlife Service 159

Marine Bird Populations of Prince William Sound, Alaska, Before and After the Exxon Valdez Oil Spill

K.K. Laing and S.P. Klowiewski, U.S. Fish and Wildlife Service 160

Intertidal

Bioavailability of Residual PAHs From the Exxon Valdez Oil Spill

Gary Shigenaka¹ and Charles B. Henry, Jr.²

¹National Oceanic and Atmospheric Administration; ²Louisiana State University 163

Impact of the Exxon Valdez Oil Spill on Intertidal Invertebrates Throughout the Oil Spill Region

Raymond C. Highsmith¹, Susan M. Saupe¹, Kenneth O. Coyle¹, Tama Rucker¹, and Wallace Erickson²

¹University of Alaska Fairbanks; ²Western EcoSystems Technology, Inc. 166

Coastal Habitat Injury Assessment: Intertidal Algal Communities

Michael S. Stekoll¹, Lawrence Deysher², and Zhanyang Guo¹

¹University of Alaska Fairbanks; ²Coastal Resources Associates, Inc. 169

Damage and Recovery Rates of Fucus in Herring Bay, Knight Island

Peter G. van Tamelen and Michael S. Stekoll, University of Alaska Fairbanks 173

Meiofaunal Recolonization Experiment with Oiled Sediments: The Harpacticoid Copepod Assemblage

J.W. Fleeger¹, M.A. Todaro¹, T.C. Shirley², and M.G. Carls³

¹Louisiana State University; ²University of Alaska Fairbanks;

³National Oceanic and Atmospheric Administration 176

Influence of the Exxon Valdez Oil Spill on Intertidal Algae:

Tests of the Effect of Residual Oil on Algal Colonization

P. Bruce Duncan¹, Anthony J. Hooten² and Ray C. Highsmith³

¹U.S. Environmental Protection Agency; ²Coastal Resources Associates, Inc.

and University of Alaska Fairbanks; ³University of Alaska Fairbanks 179

Variability of Exxon Valdez Hydrocarbon Concentrations in Mussel Bed Sediments

Patricia Rounds, Stanley Rice, Malin M. Babcock, and

Christine C. Brodersen, National Oceanic and Atmospheric Administration 182

Oiled Mussel Beds in Prince William Sound Two and Three Years After the Exxon Valdez Oil Spill

Malin Babcock¹, Gail Irvine², Stanley Rice¹, Patricia Rounds¹,

Joel Cusick², and Christine C. Brodersen¹

¹National Oceanic and Atmospheric Administration; ²National Park Service 184

Determination of Petroleum-derived Hydrocarbons in Seawater Following the Exxon Valdez Oil Spill II: Analysis of Caged Mussels <i>Jeffrey W. Short and Patricia Rounds, National Oceanic and Atmospheric Administration</i>	186
Estimation of the Exposure Concentration of the Seawater Soluble Fraction of Crude Oil from Mussel Tissue Concentrations <i>Thomas D. Mehl and Richard M. Kocan, University of Washington</i>	188
Impacts to Intertidal Invertebrates in Herring Bay, Prince William Sound, Following the Exxon Valdez Oil Spill <i>Anthony J. Hooten¹ and Raymond C. Highsmith²</i> ¹ <i>Coastal Resources Associates, Inc. and University of Alaska Fairbanks;</i> ² <i>University of Alaska Fairbanks</i>	192
Exxon Valdez Oil Spill: Recruitment on Oiled and Non-oiled Substrates <i>Anthony J. Hooten¹ and Raymond C. Highsmith², ¹Coastal Resources Associates, Inc. and University of Alaska Fairbanks; ²University of Alaska Fairbanks</i>	195
Subsistence	
The Response Process: The Goals of the Oil Spill Health Task Force <i>Thomas S. Nighswander, Alaska Native Medical Center</i>	199
Subsistence Uses of Fish and Wildlife Resources in Areas Affected by the Exxon Valdez Oil Spill <i>James A. Fall, Alaska Department of Fish and Game</i>	201
Overview of Subsistence Food Safety Testing Program <i>L. Jay Field, National Oceanic and Atmospheric Administration</i>	204
Assessment of Exposure of Subsistence Fish Species to Aromatic Compounds Following the Exxon Valdez Oil Spill <i>John E. Stein, Tom Hom, Catherine A. Wigren, Karen L. Tilbury, Sin-Lam Chan, and Usha Varanasi, National Oceanic and Atmospheric Administration</i>	206
Petroleum Hydrocarbons in Alaskan Invertebrate Subsistence Foods Following the 1989 Exxon Valdez Oil Spill <i>Donald W. Brown, Catherine A. Wigren, Douglas G. Burrows, Karen L. Tilbury, Ronald W. Pearce, Jennie L. Bolton, Susan M. Pierce, L. Jay Field, Sin-Lam Chan, and Usha Varanasi, National Oceanic and Atmospheric Administration</i>	210
Archeology/Human Impacts	
Investigations of Crude Oil Contamination in Intertidal Archaeological Sites Around the Gulf of Alaska <i>Douglas R. Reger, Alaska Department of Natural Resources</i>	215
Generating Damage Restoration Costs for Archaeological Injuries of the Exxon Valdez Oil Spill <i>Martin McAllister, Archaeological Resource Investigations</i>	219
Long-term Social Psychological Impacts of the Exxon Valdez Oil Spill <i>J. Steven Picou¹ and Duane A. Gill²</i> ¹ <i>University of South Alabama; ²Mississippi State University</i>	223
The Economic Impacts of the Exxon Valdez Oil Spill on Southcentral Alaska's Commercial Fishing Industry <i>Maurie J. Cohen, University of Pennsylvania</i>	227

Fish

Detecting Population Impacts from Oil Spills: A Comparison of Methodologies

Ray Hilborn, University of Washington 231

Coastal Habitat Studies: The Effect of the Exxon Valdez Oil Spill on Shallow Subtidal Fishes in Prince William Sound

David R. Laur¹ and Lewis Haldorson²

¹University of California Santa Barbara; ²University of Alaska Juneau 233

Survey of Oil Exposure and Effects in Subtidal Fish Following the Exxon Valdez Oil Spill: 1989-1991

Tracy K. Collier, Margaret M. Krahn, Cheryl A. Krone, Lyndal L. Johnson, Mark S. Myers, Sin-Lam Chan, and Usha Varanasi

National Oceanic and Atmospheric Administration 235

Impact of Oil Spilled from the Exxon Valdez on Survival and Growth of Dolly Varden and Cutthroat Trout in Prince William Sound, Alaska

Kelly R. Hepler, Patricia A. Hansen, and David R. Bernard

Alaska Department of Fish and Game 239

Assessment of Damage to Demersal Rockfish in Prince William Sound Following the Exxon Valdez Oil Spill

Andrew G. Hoffmann, Kelley R. Hepler, and Patricia A. Hansen

Alaska Department of Fish and Game 241

Histopathologic Analysis of Chronic Effects of the Exxon Valdez

Oil Spill on Alaska Fisheries

Gary D. Marty, Mark S. Okihiro, and David E. Hinton

University of California Davis 243

Herring

Effects of the Exxon Valdez Oil Spill on Herring Embryos and Larvae: Sublethal Assessments, 1989-1991

Jo Ellen Hose¹, Evelyn Biggs², and Timothy T. Baker³

¹Occidental College; ²Alaska Department of Fish and Game 247

Herring Embryo Stage Sensitivity to Water Soluble Fraction of Prudhoe Bay Crude Oil

Richard M. Kocan¹, Jo Ellen Hose², and Evelyn D. Biggs³

¹University of Washington; ²Occidental College; ³Alaska Department of Fish and Game 250

Egg-larval Mortality of Pacific Herring in Prince William Sound, Alaska, After the Exxon Valdez Oil Spill

Michael McGurk¹ and Evelyn D. Biggs²

¹Triton Environmental Consultants, Ltd.; ²Alaska Department of Fish and Game 254

Larval Fish Distribution and Abundance in Prince William Sound and Resurrection Bay During 1989

Brenda L. Norcross and Michele Frandsen, University of Alaska Fairbanks 256

The Prince William Sound Herring Recruitment Failure of 1989: Oil Spill or Natural Causes?

Fritz C. Funk, David W. Carlile and Timothy T. Baker

Alaska Department of Fish and Game 258

Adult Herring Reproductive Impairment Following the Exxon Valdez Oil Spill Richard M. Kocan ¹ , Timothy T. Baker ² , and Evelyn D. Biggs ¹ ¹ University of Washington; ² Alaska Department of Fish and Game	262
Summary of Known Effects of the Exxon Valdez Oil Spill on Herring in Prince William Sound, and Recommendations for Future Inquiries Evelyn D. Biggs and Timothy T. Baker, Alaska Department of Fish and Game, ...	264
Mammals	
Rehabilitating Oiled Sea Otters and Other Fur-bearing Marine Mammals: Lessons from the Exxon Valdez Oil Spill Randall W. Davis ¹ and Terrie M. Williams ² , Texas A&M University; ² NOSC Hawaii Laboratory	269
The Efficacy of the Exxon Valdez Oil Spill Sea Otter Rehabilitation Program and the Possibility of Disease Introduction into Recipient Sea Otter Populations Charles Monnett and Lisa Mignon Rotterman, Enhydra Research, Homer, Alaska	273
Clinicopathologic Alterations in Oiled Sea Otters Dying Acutely in Rehabilitation Centers A.H. Rebar ¹ , T.P. Lipscomb ² , R.K. Harris ² , and B.E. Ballachey ³ ¹ Purdue University; ² Armed Forces Institute of Pathology; ³ U.S. Fish and Wildlife Service	274
Histopathologic Lesions in Sea Otters Exposed to Crude Oil T.P. Lipscomb ¹ , R.K. Harris ¹ , R.B. Moeller ¹ , J.M. Pletcher ¹ , R.J. Haebler ² , and B.E. Ballachey ³ , ¹ Armed Forces Institute of Pathology; ² U.S. Environmental Protection Agency; ³ U.S. Fish and Wildlife Service	276
Gross Lesions in Sea Otters Found Dead Following the Exxon Valdez Oil Spill T.P. Lipscomb ¹ , R.K. Harris ¹ , and B.E. Ballachey ² ¹ Armed Forces Institute of Pathology; ² U.S. Fish and Wildlife Service	280
Age Distributions of Sea Otters Dying in Prince William Sound, Alaska After the Exxon Valdez Oil Spill D.H. Monson and Brenda E. Ballachey, U.S. Fish and Wildlife Service	282
Recovery of Sea Otter Carcasses Following the Exxon Valdez Oil Spill A. Doroff, A.R. DeGange, C. Lensink, B.E. Ballachey, James L. Bodkin, and D. Bruden, U.S. Fish and Wildlife Service	285
An Intersection Model for Estimating Sea Otter Mortality Following the Exxon Valdez Oil Spill James L. Bodkin and Mark S. Udevitz, U.S. Fish and Wildlife Service	289
Mortality of Sea Otters in Prince William Sound Following the Exxon Valdez Oil Spill Robert A. Garrott ¹ , L. Lee Eberhardt ² , and Douglas M. Burn ³ ¹ University of Wisconsin; ² Battelle Memorial Institute; ³ U.S. Fish and Wildlife Service	292
Hydrocarbon Concentrations in Tissues of Sea Otters Collected Following the Exxon Valdez Oil Spill Dan Mulcahy ¹ and B.E. Ballachey ² ¹ BioVet Services; ² U.S. Fish and Wildlife Service	293

Health, Reproduction, and Survival of Adult, Dependent, and Weanling Sea Otters in Prince William Sound From October 1989 to December 1991 <i>Lisa Mignon Rotterman and Charles Monnett, Enhydra Research</i>	296
Effects of the Exxon Valdez Oil Spill on River Otters in Prince William Sound <i>R. Terry Bowyer¹, J. Ward Testa¹, James B. Faro², and Lawrence K. Duffy¹</i> ¹ University of Alaska Fairbanks; ² Alaska Department of Fish and Game	297
Assessment of Damages to Harbor Seals Caused by the Exxon Valdez Oil Spill <i>Kathryn J. Frost and Lloyd F. Lowry, Alaska Department of Fish and Game</i>	300
Vital Rates and Pod Structure of Resident Killer Whales Following the Exxon Valdez Oil Spill <i>Craig O. Matkin¹, Marilyn E. Dahlheim², Graeme Ellis³, and Eva Saulitis¹</i> ¹ North Gulf Oceanic Society; ² National Oceanic and Atmospheric Administration; ³ Canada Department of Fisheries and Oceans	303
Assessment of Injuries to Prince William Sound Killer Whales <i>Marilyn E. Dahlheim¹, and Craig O. Matkin²</i> ¹ National Oceanic and Atmospheric Administration; ² North Gulf Oceanic Society	308
Humpback Whale Abundance and Distribution <i>Olga von Ziegesar¹ and Marilyn E. Dahlheim²</i> ¹ North Gulf Oceanic Society; ² National Oceanic and Atmospheric Administration	311
Posters	
Occupational Exposures From Oil Mist During the Exxon Valdez Spill Clean-Up <i>Carl Reller, Alaska Health Project</i>	313
Boat-based Surveys of Sea Otters (<i>Enhydra lutris</i>) in Prince William Sound, Alaska <i>Douglas M. Burn, U.S. Fish and Wildlife Service</i>	316
Hydrocarbons in Mussels and Subtidal Sediments: Graphical Presentation of Hydrocarbon Analysis Data with Geographic Map Data <i>Jeffrey W. Short, Ronald A. Heintz, and Scott Feldhausen</i> <i>National Oceanic and Atmospheric Administration</i>	319
Management of Natural Resource Damage Assessment Samples and Analytical Data <i>C.A. Maner¹, E. Robinson-Wilson², S. Korn¹, and R.L. Britten²</i> ¹ National Oceanic and Atmospheric Administration; ² U.S. Fish and Wildlife Service	320
Pre-spill and Post-spill Concentrations of Hydrocarbons in Sediments and Mussels in Prince William Sound <i>Malin Babcock, John F. Karinen, Jeffrey W. Short, and Christine C. Brodersen</i> <i>National Oceanic and Atmospheric Administration</i>	322
Surface Modeling of Floating Oil, The 1989 Exxon Valdez Oil Spill, Prince William Sound, Alaska <i>Dorothy Mortenson, Hans Buchholdt, Richard McMahon, and Randall Hall</i> <i>Alaska Department of Natural Resources</i>	324
Integration of Shoreline Oiling Data Sets <i>Randy Hall, Alaska Department of Natural Resources</i>	325
A Reconstruction of Pink Salmon Wild Stock Runs in Prince William Sound <i>William D. Templin, Jeremy S. Collie, and Terrance J. Quinn II</i> <i>University of Alaska Fairbanks</i>	328

Intertidal/Supratidal Site Selection Utilizing a Geographic Information System <i>Kimbal A. Sundberg¹, Lawrence Deysher², and Lyman McDonald³</i> ¹ Alaska Department of Fish and Game; ² Coastal Resources Associates, Inc.; ³ Western EcoSystems Technology, Inc.	330
Identification of Marbled Murrelet Nesting Habitat in Southcentral Alaska: to Guide Restoration Efforts Following the Exxon Valdez Oil Spill <i>K.J. Kuletz, N.L. Naslund, and D.K. Marks, U.S. Fish and Wildlife Service</i>	332
Homing and Straying Patterns of Coded-wire Tagged Pink Salmon in Prince William Sound <i>Daniel Sharp, Samuel Sharr, and Carol Peckham</i> <i>Alaska Department of Fish and Game</i>	335
Assessment of Intertidal Algal Populations in Prince William Sound with an Airborne Multispectral Scanner <i>Gary Borstad¹, Kimbal A. Sundberg², and Lawrence Deysher³</i> ¹ G.A. Borstad Associates Ltd.; ² Alaska Department of Fish and Game; ³ Coastal Resources Associates, Inc.	338
Stable Carbon Isotope Ratios of Prince William Sound Subtidal Sediments, Prior and Subsequent to the Exxon Valdez Oil Spill <i>A. Sathy Naidu, Stephen C. Jewett, Howard M. Feder, and Max K. Hoberg</i> <i>University of Alaska Fairbanks</i>	341
Response to the Exxon Valdez Oil Spill: A New Method to Test the Effects of Residual Oil on Intertidal Recolonization <i>P. Bruce Duncan¹, Anthony J. Hooten², and Arthur H. Weiner³</i> ¹ U.S. Environmental Protection Agency; ² Coastal Resources Associates, Inc. and University of Alaska Fairbanks; ³ Alaska Department of Natural Resources	343
The Alaska Heritage Stewardship Program <i>Debra Corbett¹ and Douglas R. Reger²</i> ¹ U.S. Fish and Wildlife Service; ² Alaska Department of Natural Resources	346
Instrumental Neutron Activation Analysis of Nesting Peregrine Falcon Feathers Collected from Prince William Sound and Norton Sound, Alaska, Breeding Populations in 1990 <i>Jimmie R. Parrish, Parrish Associates, Inc.</i>	347
Hydrocarbons in Intertidal Sediments and Mussels from Prince William Sound, Alaska, 1977-1980: Characterization and Probable Sources <i>John F. Karinen, Malin W. Babcock, Donald W. Brown, William D. MacLeod Jr., L. Scott Ramos, and Jeffrey W. Short</i> <i>National Oceanic and Atmospheric Administration</i>	348
Meiofaunal Recolonization Experiment with Oiled Sediments: Major Meiofauna Taxa <i>T.C. Shirley¹, M. Carls², J.W. Fleeger³ and N. Schizas¹</i> ¹ University of Alaska Fairbanks; ² National Oceanic and Atmospheric Administration; ³ Louisiana State University	350
Responses of Intertidal and Subtidal Meiofauna to the Prince William Sound Oil Spill <i>T.C. Shirley¹, J.W. Fleeger², C.E. O'Clair³ and S. Rice³</i> ¹ University of Alaska Fairbanks; ² National Oceanic and Atmospheric Administration; ³ Louisiana State University	354

So Why Can't Science Tell Us More About The Effects of the Exxon Valdez Oil Spill ?

Robert B. Spies

Applied Marine Sciences



The *Exxon Valdez* went aground on a reef within shouting distance of the route which carries 20% of the oil produced within the United States. It took less than 20 years for this experiment to happen. It was not planned, but it was expected, and we should have been better prepared. We can see now that we were not ready for the spill, either for clean up or for assessment of effects. We are all aware of the billions of dollars spent cleaning up the oil. I would like to give you some idea of the consequences of not being prepared to assess the damages, and at the same time provide a primer on the use of different sorts of information in injury assessment.

Any simple assessment of oil spills should recognize that the greatest risk will be to life on the surface of the water, especially animals with fur or feathers that protect them from the cold waters of a sub-Arctic environment. Since oil spilled near a coast almost always comes ashore, shorelines will be oiled, and it will persist, especially on protected beaches or in protected parts of high-energy beaches, so these environments will also be greatly affected.

We also know that floating oil will follow the prevailing winds and currents. So if we had known the tanker route, prevailing winds and currents, organisms and environments that were most vulnerable, there would have been good up-to-date information on the populations of the organisms at risk. Instead, there was virtually no usable information on the status of the intertidal and

subtidal communities in Prince William Sound. The last good census of sea otters in Prince William Sound, creatures which die from hypothermia when only 30% of their pelts are oiled, was carried out in 1984. And the last good censuses of murre colonies, the sea birds that are well known to be extremely vulnerable to oil spills, was carried out in the 1970s. I don't intend to belabor our lack of preparation for injury assessment, but will instead attempt to point out some of the consequences of not having the information. I will also indicate some of the other practical limitations to precise injury assessment.

The Natural Resource Damage Assessment studies produced results of less certainty because good up-to-date population data was not available. Consequently, additional studies were done in an effort to provide more certainty. In other words, techniques and approaches were often not closely linked to precise estimates of population level impact. This, of course, increased the costs of the Natural Resource Damage Assessment studies. Under these conditions the ultimate legal declarations or findings on injured resources become more dependent upon policy decisions regarding how uncertain results would be interpreted.

Let us start our inquiry into certainty by considering the different types of information that were gathered after the spill. These were: population counts, counts of carcasses, mortality rates of eggs and larvae, and sublethal effects

(e.g., biochemical alterations, histopathological alterations, reduced growth). Also, the presence of oil in the immediate environment of an organism or in the tissues of organisms documented exposure, the usual prerequisite for further conclusions regarding injury. This information can be ordered in a hierarchy as to seriousness of the impact and the certainty it provides.

First, let us consider population counts. Because of the great variability in populations from place to place and at different times, scientists aim to have enough pre-impact data in a variety of areas and in enough years to be able to understand how populations change naturally. This then allows a comparison of pre-impact population data to post-impact population data in both affected and non-affected areas. Generally, in order to conclude that there was an impact, the change observed in the impacted population must be larger than expected by chance compared to past fluctuations or larger than can be observed at the same time in the non-impacted population. There are statistical tests designed to test rigorously for an impact using this type of data.

Given the lack of good up-to-date baseline data on many populations, how does one assess the effect of the spill? The next best approach is usually to compare populations in oiled and unoiled areas after the spill, and this was done for most populations that were studied. This is where large amounts of uncertainty can enter into an assessment of spill impacts. It is very difficult to separate changes due to natural geographic and temporal variability from changes due to impact unless the resources are studied for many years after it can reasonably be assumed that the impact has ceased. The

confounding effects of natural variability can, however, be mitigated by replicating sites, so that several oiled areas and several unoiled areas can be matched and compared. This was done for the intertidal and subtidal studies in order to identify impacts. Still, with this approach it is possible that factors unaccounted for in the design, such as natural differences in currents, may be having systematic effects across replicate treatments.

Second, let us consider recovered carcasses. Carcass counts, usually derived from animals found on beaches, can be used to model total mortality by estimating the rates of sinking after death, rates of scavenging, burial in beaches, and the number and frequency with which beaches are searched. This was done for birds, and to some extent for sea otters. Using such modeling techniques it was estimated that the 36 thousand birds that were recovered, mainly from beaches, represented about 400 thousand total bird deaths. Carcass information will often provide the only unequivocal evidence of the impact of the spill, if mortalities are absent or if damages are within the natural variability of the population. This was the case with many species of birds, such as Bald Eagles.

Third, let us consider the role of information about elevated mortality of eggs and larvae. This applies particularly to animals or plants that produce large numbers of eggs, of which only a few percent survive, as is the case for most fish. So, for example, the spill appears to have caused an increased egg mortality in pink salmon in Prince William Sound every year since the spill, and egg and larval mortality in herring in 1989. The real biological significance of the loss of 5 to 20% of the eggs of fish that typically spawn thousands of eggs, of which on average less

than a few percent survive naturally, is a major unanswered question for environmental scientists. We put programs in place for both these species to estimate effects on adults, but no effects were detected. One could argue that there really could have been large effects due to the spill on wild stocks of adult pink salmon (the accumulated effects of increased mortality of eggs and retarded fry growth), and they went undetected because of the large variability in the counts of returning salmon. One could also argue that a change in rate of mortality in egg and juvenile stages is not going to make much of a difference in a natural mortality rate of greater than 90%. The restoration programs for pink salmon, for which we have no data that directly indicates an effect on the adult population, could amount to tens of millions of dollars. Several millions of dollars have already been spent on assessing damages to pink salmon in comprehensive programs in 1989 and 1990. The cost of herring programs for damage assessment exceeded one million dollars, but there is less that appears to be feasible for restoration of herring short of limitation of the adult harvest. These cases contrast with studies of Dolly Varden where clearly documented differences in survival of adults between oiled and unoiled streams provide a better basis for claiming injury to the population, but do not prove it without comparable natality rates.

Fourth, let us consider the role of sublethal effects. Documentation of sublethal effects of the spill may serve two general functions in an injury assessment. First, it may indicate that the exposure to oil resulted in some negative effect to the organism. Second, if the negative effect was debilitating enough it may imply decreased survival that

could, in turn, eventually effect the population. If a census is taken and there is considerable uncertainty that an organism has been impacted, the documentation of sub-lethal effects will increase the certainty. This was the case with herring, where the main impact observed was the production of deformed larvae. Conversely, if there is some doubt whether natural variability is confounding oiling effects, the lack of substantial sublethal effects suggests oil may not be the cause. These sorts of interpretations must be made cautiously, as we know very little about the toxicology of oil in the species that were studied, and most sublethal effects may have causes other than oil. For example, in Dolly Varden there are clear differences in rates of return of individuals using streams in oiled versus unoiled areas in 1989-1990 and in 1990-1991, which is strong evidence of differences in mortality rates in the populations. There were high concentrations of oil metabolites in the bile of Dolly Varden in 1989, but the concentrations decreased dramatically in 1990 without a concomitant increase in survival rate. In addition, examination of tissues of anadromous Dolly Varden revealed no differences in histopathological alterations that could be attributed to oil. It may mean that Dolly Varden are not susceptible to histopathological alterations from oil exposure, as some fish species are more resistant than others, but the outcome of these analyses does raise uncertainties about the injury.

Now let us consider how this information can all fit together. We assume that it is the population-level impact (on adults) that is of ultimate interest for injury assessment and it is the unrecovered population that deserves our highest consideration in restoration.

To return to the concept of a hierarchy of evidence, the strongest data on injury is a large number of oiled carcasses with many prime-aged animals, especially if the modeling indicates that a significant proportion of the population was killed.

Next, evidence is almost as strong if the contrast of populations after the spill shows that there was a decrease after the spill in the oiled environment that did not occur in the unoiled environment and that the decrease was probably not due to natural fluctuations (based on sufficient pre-spill information). This kind of evidence is bolstered with any of the following ancillary information: dead oiled organisms, evidence of high oil exposure, sublethal effects on survivors, or increased mortality to eggs and larvae. One case that came close to fitting this model was harbor seals, where there was a good record of the population annually through 1988 in Prince William Sound. A weaker form of this type of evidence exists where pre-spill population data were limited in extent or had been collected long enough before the spill occurred that natural fluctuations in populations could enter into the interpretation. This was common for bird and mammal populations where major data collection efforts were made in the 1970's and, in some cases, again in the middle 1980's to document populations that were subsequently affected by the spill (e.g., sea otters, murre, eagles, and many other species of birds).

Next, in cases where a population decline is based only on post-spill data which can not be contrasted with pre-spill information, evidence of spill impact is less certain. The degree of certainty can be maximized if multiple unoiled and multiple oiled sites are contrasted. As mentioned earlier, these sorts

of comparisons, without the benefit of good pre-spill information, were commonly made in the *Exxon Valdez* spill injury assessment (e.g., subtidal and intertidal communities). Careful planning of sampling sites needs to be carried out to minimize the confounding of natural differences and oil-induced differences. Also, the interpretation of differences should carefully consider the possible role of natural variability in producing the observed results. Documentation of differences in exposure, vital biological rates (including reproduction), and sublethal effects can further clarify the certainty as to whether the observed differences in post-spill populations are due to natural causes or to oil exposure.

For those of you at the symposium interested in pursuing the concept of uncertainty in injury assessment, I suggest you listen carefully to the presentations to see how the investigators deal with imperfect knowledge of injury and how oil impact and natural changes are discussed. One of the imperatives of objective science is to discuss alternative interpretations. Studies with imperfect data in which the possible role of natural causes of change are not discussed should be viewed with skepticism.

Since the consequences of being unprepared seem to be greater costs and greater uncertainty about injured resources, their recovery and need for restoration, what should we be doing now in order to be better prepared to assess damages resulting from the next oil spill? The answers seem clear in retrospect—ongoing monitoring programs collecting data on intertidal and subtidal zones; annual counts of sea otters, eagles, murre and other sea birds; and gathering more experimental information on oil toxicology of common species. A basic and rela-

tively inexpensive monitoring program carried out over many years might tell us enough so that if another spill were to occur along an Alaskan tanker route we might get better injury information at lower cost. In the process we would also learn more about the natural resources we are trying to protect.



Fate of the Oil Spilled from the T/V Exxon Valdez in Prince William Sound, Alaska

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This paper integrates field observations from several different investigations, along with published experimental and field data on spilled oil in the marine environment, in an effort to reconstruct the overall fate of the 10.8 million gallons of oil spilled from the Exxon Valdez through the fall of 1992. During the first 6 weeks after the spill, the location and movements of the floating oil were tracked visually by observers from aircraft and on the shoreline. For this period, the distribution of floating and beached oil was hindcast by NOAA's Oil Spill Simulation Model (Galt et al. 1991), which took into account real-time data for the wind fields and currents moving the slick. A weathering model based on extensive experimental data (Payne et al. 1991) was used to estimate initial losses due to evaporation and related weathering processes. By the end of April, 1989, approximately 20% of the spilled oil had evaporated, and approximately 20-25% had been dispersed naturally into the water column. About 25% of the oil had been carried out of Prince William Sound as floating or dispersed oil, and most of the remainder (approximately 40-45%) had beached within the Sound. No quantitative measurements were made of volatile constituents in the atmosphere, and the oil evaporation rates were estimated

from volatility models based on theoretical data under the temperature and wind conditions prevailing at spill time. The accuracy of theoretical evaporation rates was confirmed through compositional analyses of floating oil. During the first 3 days of the spill, before its initial landfalls, the most volatile constituents (accounting for approximately 13% of the spill volume and including most of the benzene, toluene, and alkanes through C₁₀) evaporated (Payne et al. 1991). The atmospheric trajectory for these initially evaporated materials from the still concentrated slick passed over Naked Island and Eleanor Island (Hanna et al. 1991). Measurement of the oil dissolved and/or dispersed in the water column was initiated 6 days after the spill. Concentrations of volatile, monoaromatic hydrocarbons in seawater were often in the 1-5 ppb range in close proximity to floating or beached oil during early April, but had generally declined to concentrations near or below analytical detection limits (about 0.18 ppb) by the end of April (Neff 1991). During the first 2 weeks of April, concentrations of total polynuclear aromatic hydrocarbons (PAH) in the water column were frequently also in the range of 1-7 ppb, especially near heavily oiled beaches (Neff 1991; Payne et al. 1991; Short &

Rounds 1993a). Aqueous PAH concentrations were generally below 1 ppb by May 1, and below 0.1 ppb after July 1. In some locations, however, elevated petroleum hydrocarbon concentrations were measured in transplanted filter-feeding mussels into 1991, indicating the persistence of extremely low concentrations of dissolved and particulate oil components in the water column through that time (Short & Rounds 1993b).

The oil recovered by skimming operations in 1989 accounted for about 8.5% of the original spill volume. Cleanup operations on the beaches during the first four summers led to the recovery and disposal of approximately 31,000 tons of solid oily wastes (Carpenter et al. 1991; ADEC 1992), which were estimated to account for 5 to 8% of the original spill volume. Annual beach surveys conducted in the fall and spring indicated that about 90% of the oil in surface (<25 cm) beach sediments was removed by natural processes (storm erosion and biodegradation) during winter 89-90, whereas only about 40% of the deeper oil was removed (Michel et al. 1991, Michel & Hayes 1992). By 1992, the combination of natural processes and cleanup activities had eliminated nearly all of the surface oil, though small amounts persisted along many shoreline segments in the Sound. Reworking by storms and berm relocation had removed nearly all the oil from the upper intertidal zone. In a few areas appreciable quantities of oil remained trapped under the central platforms of exposed cobble/boulder beaches, where it had penetrated deep into the substrate. The oil trapped in such protective crevices may be only moderately weathered, unlike oil deposited onto coarse sand or fine sediments, which by 1992 was generally highly

weathered (Roberts et al. 1993). Relatively unweathered oil also persisted in 1992 in the sediments and byssal mats underlying some low gradient mussel beds in Prince William Sound (Babcock et al. 1993). Coincident with the decline of oil on the beaches, concentrations of oil (O'Clair et al. 1993) and numbers and activity of hydrocarbon-degrading microbes (Braddock et al. 1993) increased in subtidal sediments in 1990. At heavily oiled locations (eg. Northwest Bay on Eleanor Island, Herring Bay on Knight Island, and Sleepy Bay on Latouche Island), peak subtidal concentrations occurred between September 1989 and September 1990 at depths between 3 and 20 m. Corresponding decreases were not always noted in oil concentrations at mean lower low water (0 m). Near some heavily oiled areas, low concentrations of residual petroleum hydrocarbons and associated concentrations of microbial hydrocarbon-degradation also were detectable during the summer of 1990 in deeper (40-100 m) subtidal sediments where no activity had been detected in 1989. Measures of microbial hydrocarbon degradation had declined in 1991 at all sites and depths. Sediment traps deployed at 10-20 m depths in these areas indicated that oil was sorbed onto suspended particulate material (Sale et al. 1993), reflecting transport of oiled sediments from the intertidal zone into deeper waters, at least through the winters of 1989-90 and 1990-91.

Much of the floating mousse that departed Prince William Sound was deposited on shorelines in the Kenai and Kodiak areas. Galt et al. (1991) estimated that about 2% of the spilled oil floated past Cape Douglas into the Shelikof Strait where heavy mousse was deposited between Hallo Bay and Wide Bay during

April 29-May 2, 1989. Because of the extent of prior weathering and emulsification, this stranded mousse did not penetrate into shorelines nearly to the extent that the fresh oil did in Prince William Sound, and as a result, was much more amenable to physical removal. In the areas of heaviest shoreline oiling along the coasts of the Kenai and Alaska peninsulas, extensive cleanup activities were carried out during June-July 1989, and by early August, most of the beaches appeared generally clean with only sparsely distributed small tarballs and mousse patties. On sandy beaches some mousse was buried by shifting sands (Dewhurst 1993a). In fall 1989, after initial cleanup operations, a total of 9.7 km of shoreline along the Kenai coast was characterized as heavily oiled and 12.9 km as moderately oiled. Of this total, 4.35 km and 1.0 km were still described as moderately oiled in fall 1990 and spring 1991, respectively (ADEC 1992). By summer 1990, the most commonly observed form of oil along the coast of the Becharof Refuge on the Alaska Peninsula was staining on shoreline debris (Dewhurst 1993b).

Estimates will be provided for the distribution of the spilled oil in different environmental compartments from the start of the spill through the fall of 1992, and the chemical evolution, or weathering, of the oil during that time. Evaporation was a dominant process during the first few days. Relatively unweathered oil stranded along shorelines in Prince William Sound, where it penetrated into beach gravel. The oil that exited the Sound was more highly weathered and generally did not penetrate deeply into the substrate when grounded. Most of the oil beached in the Sound has been removed through a combination of biodegradation, natural erosion and cleanup

activities, but relatively unweathered oil still remained in 1992 in isolated and protected situations in the sediments of some Prince William Sound beaches. The overall behavior of the spilled oil from the *Exxon Valdez* was generally consistent with prior understanding of the behavior and fate of oil in the marine environment, and the observations and measurements recorded after this spill provided useful validation for existing models of oil transport and weathering.

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The Exxon Valdez Cleanup—The First Six Months

VADM Clyde E. Robbins

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Vice Admiral Robbins was the Federal On Scene Coordinator for the *Exxon Valdez* oil spill from April 16 until September 30, 1989. It was during this period that the Exxon Company undertook this country's largest oil spill cleanup. With over 12,000 personnel involved, it was described by some as rivaling the Normandy invasion.

From the moment of the grounding of the *Exxon Valdez*, it was inevitable that a great proportion of the 260,000 barrels of North Slope crude would find its way onto the pristine shores of Alaska, causing changes to the lives and livelihoods of thousands. Alaskans who had never worked outside of the home found themselves working along the shorelines, attempting to minimize the impact of such a spill. Fishing, hunting and recreation came to a standstill for the summer of 1989. Fishing boats were pressed into service as support vessels for the monumental task. New buildings were built in Valdez and other towns and villages; telephone services were increased many fold; millions of dollars worth of equipment was imported; communities swelled to many times their original size; barges, boats, cranes, and house trailers grew like topsy. No effort was spared.

Despite all this, cleanup was slow and marginally effective. Oil on any shoreline is tough to remove, but Alaska's presented new challenges. New methods and equipment were aggressively tested—many were found wanting, while others were breakthroughs. The mechanics of the cleanup operation were

only a portion of the problems faced by the Federal On Scene Coordinator.

Complicating the cleanup process were problems which made decision-making difficult. Everyone, it seemed, had an interest in the operation. Some of those interests were based on a true concern for the environment, while others appeared to have a hidden agenda. Faced with an operation which dwarfed any past effort, every action by the participants was magnified under the microscopic eye of the news media. Little escaped the watchful eye of the hundreds of agencies, departments, civil organizations, environmental groups, and others who had an interest. One official described it as a frustrating effort to keep "all the kittens in their box".

The national document which organizes the efforts to prevent and to clean up oil spills, the National Contingency Plan, by its very nature, is a consensus document. To win agreement on its content, compromises were necessary. As an example, the Federal On Scene Coordinator is just that; a coordinator without real authority. The State On Scene Coordinator is in the same position. If federal interests differed with Alaska's, conflict often ensued. To counter some of the conflict, committees were formed to provide advice to the Federal On Scene Coordinator. These provided a high level of interest group input into decision-making and were quite successful.

The weakness in the National Contingency Plan did not greatly hamper the cleanup operation—it only complicated

the process. There were exceptions, however. An example was the refuse issue. Very early in the cleanup it was apparent that a means of getting rid of refuse was going to be needed. With tons and tons of trash being generated each day, local facilities in Valdez and other small towns were not going to be adequate. In a joint meeting on the issue it was decided to incinerate the oil tainted debris. Exxon immediately contracted for incinerators to be constructed and barged to Alaska. At that point the Federal On Scene Coordinator lost control. Public resistance to incineration grew despite some pretty convincing scientific information which indicated that no one was going to be endangered by the process.

The conflict raged throughout the summer with the final result that one of the two incinerators was used, but for less than a fortnight. Most of the refuse was hauled to Oregon for disposal in a very expensive landfill reserved for hazardous materials. More than \$5 million was wasted on this project.

Another problem was generated by the concern of Alaskans who feared that Exxon would leave the mess at the end of the summer, never to return. The state therefore wanted to ensure that every shoreline was completely free of oil before the work parties moved on. With the general agreement that work would have to terminate at the latest on 15 September in order to avoid the dangers of winter, complete cleanup was an impossible task.

The Federal On Scene Coordinator set as the major goal for the summer of '89 to render the polluted shoreline relatively neutral so the oil wouldn't spread during the winter months, causing more damage. Tensions ensued between the Federal On Scene Coordinator and the

State representatives over such matters as "how clean is clean" and when to move on to the next shoreline. Eventually a compromise was reached. It was agreed to use the word "treated" rather than "cleaned".

These are two examples of the problems which arose during the cleanup operations, but there were successes. Through Exxon's worldwide organization, equipment was made available the likes of which had never been seen before. Skimmers, water heaters, fertilizers and techniques were brought to bear effectively without concern for the cost involved. It was found for instance, that the French had some of the best skimmers available. Those skimmers dotted the shoreline as the summer passed.

It was found that the French had developed a fertilizer which was ideal for bioremediation applications. Tested by a joint program with the U.S. Environmental Protection Agency and Exxon, it was found that there were microbes existing along the shorelines in Alaska which could help in the cleanup with little negative impact on the environment. These critters only had to be encouraged with some nutrients and the French fertilizer was the best available.

Unfortunately 450 tons were needed and only some 250 tons were available. The French factory which made it was to be shut down for August vacations. Exxon used their considerable resources to induce the plant to stay open until the full 450 tons were made.

The application of that fertilizer to the shorelines to enhance the growth of indigenous microbes which consume the oil was one of the great successes. Shorelines that were black with residual oil after "treatment," were greatly improved after application of the fertilizer.

There were other successes. While fishing came to a standstill during that infamous summer of '89, the salmon were largely unscathed. Hatcheries were protected and later years' runs were very healthy. Shorelines which experts said would take decades to recover, seem to have largely recovered in under four years. Only time will tell what the long term impact of the *Exxon Valdez* spill will

be, but there is optimism.

At the end of the summer, those who were there could honestly say that they gave it "their best shot". Perhaps we'll never know whether the long hours of dedication of those thousands of people significantly helped save the environment, but if they didn't, it wasn't for the lack of trying. One lesson came through to everyone—spilling oil has to stop!

Cultural Resources and the Exxon Valdez Oil Spill

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The state of knowledge about human history in the Northern Gulf of Alaska at the time of the *Exxon Valdez* oil spill was fragmentary at best. Except for excavations at a single site, no substantive archaeological work had been done in Prince William Sound since the 1930's. No serious large scale inventory or testing had ever been done on the Outer Coast of the Kenai Peninsula. Kodiak Island and the Alaska Peninsula were the only areas known archaeologically and data came from a small number of sites. In the face of impending danger to sites, resource managers in the spill area were forced into the awkward position of protecting known sites while at the same time scrambling to identify undocumented sites.

Archaeologists were alarmed about potential injuries to sites in the spill area because many sites had subsided into the intertidal zone during the 1964 earthquake. Those sites would be located in the path of the spreading oil slick. The primary concern was the effect of crude oil and cleanup procedures on the sites. Coupled with that was the concern that undocumented sites could unknowingly be impacted during cleanup. The unexplored status of the area made the problem particularly disturbing.

The charge of protecting cultural resources during the spill and subsequent cleanup came from several authorities. Federal mandates included the Archaeological Resources Protection Act (ARPA) and the National Historic Preservation Act of 1966 (NHPA). The ARPA directed

land managing federal agencies to protect cultural sites on land they managed. The NHPA directs federal agencies "undertaking actions" such as spill cleanup, to consider the effects on cultural resources. The Alaska Historic Preservation Act protects sites on state land, including the intertidal zone.

Close coordination between the Alaska State Historic Preservation Office (SHPO), the U.S. Coast Guard On-Scene Coordinator, native representatives, federal representatives and the Exxon Cultural Resource Program protected cultural resources during cleanup. Coordination occurred during cleanup planning by Exxon and a representative of the SHPO reviewing Exxon archaeologist's reports for adequacy and planned activities. Proposed cleanup methods and scope were altered prior to initiation to reduce danger to known sites. Federal and native representatives reviewed and discussed actions during Shoreline Committee meetings. Agency and native corporation archaeologists monitored cleanup activities on a sporadic basis.

Agency concerns about the effect of crude oil contamination on the radiocarbon dating process prompted a laboratory study of those effects. The contractor concluded that significant effect would occur but that sample cleaning partially reduced the error. Testing that conclusion on specific sites was the next step of injury assessment. Unfortunately, the radiocarbon study did not get funded until 18 months after the spill and follow-

up studies were delayed another year.

Due to delays, field damage assessment studies did not take place until 1991. Damage assessment studies were requested during 1989 by the panel of archaeologists advising the Trustees staff. In 1990 the C-14 laboratory study was funded. The state and federal agencies provided funds during 1991 and the State University of New York, Binghamton, began fieldwork at the end of the 1991 season. The scope of the 1991 project was reduced when Trustee representatives decreased available funding. The project aimed at testing a series of sites for injury, surveying to check the adequacy of the Exxon surveys, checking several soil chemicals for alteration by oiling, and testing a model developed to predict site locations. Radiocarbon dating tested sites was a basic part of the project.

Delays in awarding a damage assessment contract during 1990 and 1991 prompted the state to begin a much smaller and tightly focused study in May 1991. The state study followed up on the conclusions of the earlier radiocarbon study by checking selected sites. Alternate methods of dating archaeological remains were contrasted with radiocarbon estimates to check for injuries from oil contamination. The study was designed to compliment the larger multi-agency study.

Another approach to assessing injuries to archaeological sites by spill related activities was a compilation of information in agency and Exxon documentation. Field notes from various parties, monitoring reports, and SCAT reports were examined to determine kinds and degree of impacts. This method of investigation was used partly because of the need for a more timely idea of impacts and partly to assess the adequacy

of the documentation process. Although results of the document study provided only a rough idea of injuries, it was very useful in estimating injuries resulting from cleanup and attendant vandalism. Valuable insights were gained in the types and detail of documentation needed from site identification surveys and monitoring. The study also provided the core data base for assessing damage to the area sites.

Assessment of monetary values for restitution followed the documents study. Findings of the state and multi-agency funded studies also provided data useful in determining levels of damage. A method of assessing damage based on ARPA procedures was used to give a monetary estimate. The method has been repeatedly used in other cases and provided very conservative cost estimates. The most clearly injured sites were assessed for damages while the less adequately documented sites were deleted from the process. Sites considered had sustained damage from cleanup related activities rather than from oiling.

Several useful conclusions about archaeology and the oil spill are possible based on damage assessment studies. The most important conclusion was that the sites generally were not directly effected by the spill. The most extensive damage resulted from vandalism because of more people having widespread knowledge about area sites. That knowledge expanded due to increased population and activity levels during the cleanup. Another source of impact was directly from cleanup activities. Impact from cleanup was slight because of cleanup workplan review and clean-up technique alteration with archaeology in mind. The Exxon Cultural Resource Program/agency cooperation was effective

in keeping cleanup impacts minimal.

Lack of field assessment studies for the first two years after the spill caused inefficient resource management. Information collected by archaeologists and other monitors early in the cleanup phase could have included the data needed to protect the resource. Future considerations identified from the experience of

the *Exxon Valdez* oil spill are the need for a credible basic inventory, a response plan, and improved monitoring. Inventory and planning are long term projects which are typically difficult to fund. Improved monitoring will require analysis of existing studies and listing the information required during future emergencies.



Subsistence

James A. Fall

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This plenary session presentation provides information about changes in subsistence uses of fish and wildlife resources in 15 Alaska Native communities whose hunting, fishing, and gathering areas were affected by the Exxon Valdez Oil Spill. It also reviews the work of the Oil Spill Health Task Force, a group of agencies and organizations which formed to address questions concerning the safety of using subsistence foods harvested from the oil spill area.

Before 1989, the Division of Subsistence of the Alaska Department of Fish and Game conducted baseline research in the 15 communities. These are Tatitlek and Chenega Bay in Prince William Sound; English Bay (Nanwalek) and Port Graham in lower Cook Inlet; Akhiok, Karluk, Larsen Bay, Old Harbor, Ouzinkie, and Port Lions in the Kodiak Island Borough; and Chignik, Chignik Lagoon, Chignik Lake, Perryville, and Ivanof Bay on the Alaska Peninsula. In 1990, the population of these 15 communities was 2,036, 82.3 percent of which was Alaska Native.

These studies documented large and diverse subsistence harvests in the 1980s, ranging from about 200 pounds per person to over 600 pounds per person usable weight per year. These are substantial harvests, given that the average American family purchases about 222 pounds per person of meat, fish, and poultry annually. These subsistence harvests contained a wide variety of resources, including salmon and other fish, marine invertebrates, land mammals, marine

mammals, birds and eggs, and wild plants. Virtually every household in all 15 villages used and harvested wild foods, which were widely shared within and between communities. A patterned seasonal round of subsistence harvesting structured much of economic, social, and cultural activities in each community.

In early 1990, Division of Subsistence researchers interviewed representatives of 403 households in these 15 communities. Study findings revealed that after the spill, subsistence harvests declined greatly in the 10 communities of Prince William Sound, lower Cook Inlet, and the Kodiak Island Borough compared to pre-spill averages. Annual per capita harvests in Chenega Bay and Tatitlek were down 57 percent. In these villages, the range of resources used also dropped in the 12 months after the spill. While the average household in Tatitlek used 22.6 kinds of wild foods from April 1988 through March 1989, in the next year, this average was only 11.6 types. The change at Chenega Bay was much like that of Tatitlek. In a 12 month study year in 1985-86, the average household at Chenega Bay used 19 kinds of wild foods, compared to just 8.2 kinds in the year after the spill. Similar changes were documented for English Bay and Port Graham. Also, subsistence harvests in all six Kodiak area villages declined in 1989 compared to pre-spill averages, although a wider range of changes was documented. In contrast, subsistence harvests in the five Alaska Peninsula com-

munities showed little change, or increased, in 1989 compared to 1984, the only pre-spill year for which comprehensive data are available.

When asked to provide reasons for these declines, 33.2 percent of the sampled households attributed reductions in overall subsistence harvests to concerns about resource contamination, and 44 percent said such a concern had caused a reduction in their harvest of at least one kind of subsistence food. Levels of concern about contamination were notably higher among Prince William Sound (92.1 percent) and Lower Cook Inlet (77.8 percent) households than in the Kodiak area (29.5 percent) or Alaska Peninsula (22.8 percent) communities. Other reasons cited for lowered levels of subsistence uses included the time harvesters spent on the oil spill cleanup and the perception that less resources were available because of spill-induced mortalities.

As noted above, the primary response to the issue of subsistence foods contamination was directed by the Oil Spill Health Task Force. The Task Force included the Indian Health Service; the Alaska departments of Fish and Game, Health and Social Services, and Environmental Conservation; the National Oceanic and Atmospheric Administration; Exxon; and two regional Alaska Native service organizations, the North Pacific Rim (now known as Chugachmiut) and the Kodiak Area Native Association. The task force coordinated and reviewed research on subsistence food safety, developed a consensus on health issues, and communicated findings of the studies to the communities through health bulletins, newsletters, public meetings, and a video.

As part of this effort, 309 samples of fish and 1,080 samples of shellfish that had been collected from 146 traditional

harvest areas near the 15 communities over a three year period were tested for signs of oil contamination. For the most part, the samples were collected with the assistance of residents of the communities at sites recommended by community leaders. The tests were conducted at the National Marine Fisheries Services' Northwest Fisheries Center in Seattle. These highly sensitive tests measure the concentrations of polycyclic aromatic hydrocarbons (PAHs) in edible tissues. The findings were reviewed by a panel of toxicologists who provided a health assessment.

Evidence of exposure to oil was found in some of the samples of fish, but none of the samples had PAH concentrations so high as to be a human health concern, according to the group of toxicologists. The majority of shellfish from most sites also were determined to be safe to eat. However, even in 1991, PAH concentrations remained elevated in shellfish samples from Windy Bay, a highly oiled site on the lower Kenai Peninsula. Therefore, the advice of the Task Force has remained cautious; fish from the oil spill area are safe to eat, but people should still avoid using marine invertebrates from beaches with obvious oiling.

As part of the Oil Spill Health Task Force program, samples of ducks, deer, and marine mammals were also tested. Evidence of exposure to oil was found in some of the duck and deer samples, but PAH concentrations were well below those considered dangerous by the health experts. The blubber from oiled seals taken in Prince William Sound soon after the spill had elevated PAH concentrations. Seals taken in the same areas a year later, which no longer showed obvious signs of oiling, showed reduced concentrations of PAHs in their blubber, al-

though these concentrations were still higher than samples from seals from unoiled areas. Members of the Task Force advised that these elevated concentrations of PAHs in ducks, deer, and marine mammals were still well below those considered to be of concern for subsistence users.

In 1991, the Division of Subsistence conducted follow-up interviews with 221 households in seven spill-area villages pertaining to subsistence harvests during the second post-spill year. Harvest levels increased at Port Graham, Larsen Bay, and Karluk, and matched at least one pre-spill measurement. The range of resources used was also up substantially in all three communities. In two other communities, Ouzinkie and English Bay, harvest levels also increased, but remained below pre-spill averages. This general increase in harvest levels and range of wild foods used suggests some renewed confidence in using subsistence foods during 1990. On the other hand, lingering concerns about food safety were expressed in all five villages. Some families reported that they had resumed their subsistence harvests despite misgivings because they could not afford to purchase substitutes and could no longer do without culturally important foods.

In contrast, no evidence of a recovery in subsistence uses in the second post-spill year was found for Chenega Bay and Tatitlek. At Chenega Bay, subsis-

tence harvests from April 1990 through March 1991 were 139.2 pounds per person, virtually the same as the previous year (148.1 pounds per person) and still well below the pre-spill average of 340 pounds per person. At Tatitlek, the 1990-91 per capita harvest was 152.0 pounds, compared to 214.8 pounds per person in the first post-spill year and a pre-spill average of 497.6 pounds per person. In these Prince William Sound communities, deep concerns about the safety of using subsistence foods from their traditional harvest areas continued.

In addition, respondents from Chenega Bay and Tatitlek reported perceived declines in the numbers of some important subsistence resources, such as certain species of waterfowl, marine invertebrates, and marine mammals, which led to well below normal subsistence harvests in 1990-1991.

Preliminary information collected by the Division of Subsistence during another round of systematic surveys suggests that subsistence harvest levels in the Prince William Sound communities increased in the third year after the spill. However, subsistence harvests of waterfowl, marine invertebrates, and marine mammals continue to be abnormally low. Also, despite increased harvests and uses, concerns persist about the long-term health risks associated with using subsistence foods.

Overview of Intertidal Processes, Damages and Recovery

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The intertidal zone occupies the unique triple interface among the lithosphere, hydrosphere, and atmosphere. The land provides the substrate for occupation by intertidal organisms, the water the vehicle to supply necessary nutrients and to transport propagules (newly colonizing organisms), and the air a medium for passage of solar energy and also a source of physiological stresses. The intertidal zone is exceptionally biologically active and productive. Wind and tidal energy combine to supply the intertidal zone with planktonic foods produced in the productive photic zone (uppermost layer of water where photosynthesis occurs) of the coastal ocean. Runoff from the adjacent land mass injects new supplies of inorganic nutrients to fuel high coastal primary production of plants. The consequent abundance and diversity of life and life forms in the intertidal zone serves many consumers, coming from land, sea, and air, and including humans. The intertidal ecosystem simultaneously provides intrinsic aesthetic and cultural appeal in an environment undeniably shaped by forces outside the control of mankind.

The same physical transport processes that are responsible for its high level of biological production also place the intertidal zone at great risk to floating pollutants, such as oil. Oil floats on the surface of the water, is transported there by winds and currents, and, if spilled near the coast, is likely to be deposited on shore. There it encounters intertidal organisms and adheres to intertidal substrata, where it remains until chemical

transformation and physical transport remove it. Thus, the intertidal zone becomes a natural focal point for the study of damages to natural resources following a coastal oil spill.

The biota of intertidal habitats varies with changes in physical substrate type, wave energy regime, and atmospheric climate. Substrata range from immobile rocks, to boulders and cobbles, to sands, and finally to muds at the finest end of the size spectrum. Rock surfaces in the intertidal zone tend to be populated by epibiota (surface dwelling organisms), attached macro- and microalgae, sessile suspension-feeding invertebrates, and mobile grazing invertebrates. Unconsolidated ("soft") substrata, the sands and muds, are occupied by large plants in low wave energy areas, and microalgae and infaunal (buried) invertebrates in all energy regimes. Mobile scavenging and predatory invertebrates occur on both high and low energy shorelines. Intertidal communities vary with wave energy because of biomechanical constraints (especially on potentially significant predators), changing levels of food supply, and interdependencies between wave energy and substratum type. Intertidal communities tend to be more luxurious in temperate climates, whereas freezing and ice scour limit intertidal biota in polar regions and desiccation and effects of intense solar radiation may limit intertidal biota in the tropics. The rocky intertidal communities of the Pacific Northwest may be the richest on earth.

The rocky intertidal ecosystem probably represents the best understood natural community of plants and animals anywhere. Ecologists realized over 30 years ago that this system was uniquely well suited to experimentation because the habitat was accessible and the organisms moveable and manipulable.

Consequently, ecological science has produced a detailed understanding of the complex of processes involved in determining patterns of distribution and abundance of rocky intertidal organisms. Plants and animals of temperate rocky shores exhibit strong patterns of vertical zonation in the intertidal zone. Physical stresses tend to limit the upper distributions of species populations and to be most important high on shore, whereas competition for space and predation tend to limit distributions lower on the shore. Surface space for attachment is potentially limiting to both plants and animals in the rocky intertidal zone. In the absence of disturbance, space becomes limiting and competition for that limited resource results in competitive exclusion of inferior competitors and monopolization of space by competitive dominant species.

Physical and biological disturbance and recruitment limitation are all processes that can serve to maintain population densities below the level at which competitive exclusion occurs. Because of the importance of such strong biological interactions in determining community structure and dynamics in this system, changes in abundance of certain species can produce intense direct and indirect effects that cascade through the ecosystem.

Intertidal communities are open to use by consumers from other systems: the great extent and importance of this

habitat as a feeding grounds for major marine, terrestrial, and aerial predators render the intertidal system a key to integrating the health and recovery of the entire natural ecosystem. The intertidal resources of Prince William Sound and adjacent areas affected by the *Exxon Valdez* oil spill, for example, are critically important feeding grounds for many marine consumers (sea otters, Dungeness and other crabs, juvenile shrimps, rockfish, cod and juvenile fishes of other exploited species), terrestrial consumers (bears, river otters, and people), and avian consumers (black oystercatchers, harlequin and other ducks, and numerous shorebirds). Thus, the intertidal habitat provides vital ecosystem services in the form of prey resources for all coastal habitats, as well as human services in the form of commercial, recreational, and subsistence harvest of shellfishes and aesthetic, cultural, and recreational opportunities. The intense focus on the intertidal habitat following *Exxon Valdez* oil spill was well justified.

Rapidly following the spill, oil became deposited along hundreds of miles of intertidal habitat in Prince William Sound, the Kenai Peninsula, lower Cook Inlet, the Alaskan Peninsula, Kodiak Island and beyond. A massive cleanup process was mobilized with the intent of displacing the oil from the intertidal zone where it was so evident to even casual observers and where its mere presence degraded recreational and cultural values of wilderness shorelines. Cleanup techniques were diverse and not chosen to minimize damages or maximize recovery rates of intertidal organisms. In fact, evidence from subsequent biological studies suggests that the widely applied pressurized hot water treatment was more damaging than the oil to inter-

tidal ecosystems and retarded recovery. The cleanup efforts did serve to remove much of the deposited oil from the intertidal habitat and to displace it out of sight into the subtidal environment. Only very small areas of salt marsh habitat were oiled, where previous studies have shown natural cleanup of the low-energy, anoxic muds to take exceptionally long times. Winter storms accelerated the removal of oil from heavily oiled beaches, with higher rates of cleaning at more exposed beaches.

Despite the intense cleanup effort and the natural degradation and physical redistribution of residual hydrocarbons on intertidal beaches, oil in the intertidal zone remains a major problem more than 3 years later. Armored from wave action by surface layers of cobbles and by overlying mussels, large pockets of largely unweathered oil exist in the protected interstices among the rocks of intertidal shorelines in widely scattered areas. This oil is slowly leaking hydrocarbons into the water, where it is maintaining high levels of contamination in the tissues of the closely associated suspension feeders, such as the blue mussels.

Because of the nature of the blue mussel as "the universal prey" for so many consumers (including especially sea otters, river otters, harlequin ducks and black oystercatchers), the likelihood of continued food chain contamination is high. Continuing reproductive problems with each of these species in the oil spill area may be related to this ongoing contamination of a major component of their diet. In addition, much oil remains in the intertidal zone as asphalt pavement relatively high on shore, where it alters the rock surface chemistry, effects local thermal change in the habitat, and persists as a visual blemish to the previ-

ously pristine shorelines.

Intertidal suspension-feeding invertebrates represent a group of special importance as monitors of contamination of the marine environment. These organisms filter huge volumes of seawater to collect food and thereby can concentrate biologically available pollutants (metals and organic contaminants), vastly enhancing detection capabilities over what is possible in water analyses. A long-term monitoring of hydrocarbon levels by National Oceanic and Atmospheric Administration staff in Prince William Sound using blue mussels predated the *Exxon Valdez* spill and provided background information, against which the large increase of hydrocarbon contamination from the 1989 spill is clearly evident. This pollution-monitoring tool has also shown the subsequent decline in hydrocarbon concentrations to background levels in most places except where the contaminated mussel beds persist.

Analysis of several species of clams by the Alaska Department of Fish and Game also revealed the magnitude and extent of contamination of the food chain with hydrocarbons from the spill, using clams as another intertidal suspension feeder with tremendous importance as a prey resource for higher consumers and as a subsistence food. Clams and especially mussels were sampled intensively over broad geographic areas extending beyond Prince William Sound to provide a relatively complete spatio-temporal tableau of the pattern of *Exxon Valdez* oil spill contamination using a single method of ecological relevance.

The affects of the *Exxon Valdez* oil spill on the rocky intertidal communities of plants and animals has been studied intensively and extensively by special-

ists at the University of Alaska, NOAA, and the U.S. Environmental Protection Agency. These studies are distinguished by their broad geographic coverage (which includes the Kenai-lower Cook Inlet and the Kodiak-Alaskan Peninsula areas as well as Prince William Sound), their statistically sound sampling schemes, and by their focus on the mechanistic interactions among species populations that control community response and recovery.

The *Exxon Valdez* oil spill caused widespread and intense damage to rockweed (*Fucus*) in the high and mid intertidal zones. Recovery has been slow, in part because this seaweed recruits new plants most effectively within the microenvironment provided by shading and moisture under its own canopy. *Fucus* is the major provider of structural habitat in the Alaskan intertidal zone, so the indirect impacts of the loss and slow recovery of *Fucus* cover are great and may explain several responses in intertidal invertebrate populations.

The intertidal invertebrates exhibited several large declines and some related increases in abundance in response to the *Exxon Valdez* oil spill. Species that

were affected include limpets, barnacles, and probably various grazers such as snails. Responses of the intertidal invertebrate communities were not identical in all geographic areas, even within the sheltered rocky shore habitat, but all geographic regions exhibited significant impacts.

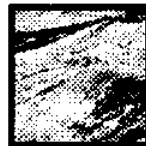
Application of pressurized hot water during cleanup of the oil was at least as damaging as the oil itself to intertidal invertebrates and delayed biological recovery. Natural recovery has varied with elevation: the recovery of the high intertidal zone has only recently been initiated, whereas recovery has progressed substantially in the lower zones.

Perhaps the most intriguing yet disturbing outcome of the study of rocky intertidal community response to the oil spill is the demonstration of delayed, indirect effects appearing more than two years after the spill. In a system characterized by strong interactions, such indirect effects are not surprising, and they emphasize the importance of understanding the nexus of interactions among populations involved in response and recovery of marine ecosystems following perturbation.

Subtidal Oil Contamination and Biological Impacts

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The *Exxon Valdez* oil spill provided a unique opportunity to examine the fate of oil contamination below the water surface, and to study impacts to species and communities in an environment where assessment was generally not confounded by prior pollution. Although subtidal chemical and biological baseline studies prior to the spill were generally absent, comparison between oiled and non-oiled habitats was possible. This paper summarizes the studies measuring oil contamination and biological impacts below the water surface; a habitat which provides a valuable and varied resource base that supports many harvestable species and their prey.

Oil contaminated sediments that are submerged are more difficult to evaluate than intertidal or supratidal sediments. Sediments that are not submerged are often heavily oiled due to direct exposure to oil on the sea-surface resulting from the spill, so that the presence of oil may be visually evident, and biological impacts obvious. In contrast, sub-surface oil is difficult to monitor, contamination is seldom at a concentration level that can be observed visually, and the biological impacts are not as obvious as are the corpses of otters or birds.

On March 24, 1989, approximately 11 million gallons of North Slope crude oil was released into Prince William Sound. In the first few weeks after the spill, about 50% of the oil evaporated or exited Prince William Sound, leaving the remaining 50% within the sound and mostly stranded on beaches (Wolfe et al.

1993). After the first month, the oil on and within the beaches became the reservoir for contaminating adjacent water and subtidal sediments, either through cleaning activities or natural weathering. It is not known precisely how much of the oil entered the water, as soluble compounds or as particulates, nor it is known how much was carried by various mechanisms to submerged sediments.

Low quantities of oil (1-8 ppb total aromatic hydrocarbons) were measured in the water column (Short and Rounds 1993a) in the first two weeks after the spill, when large quantities of oil were floating and moving through the Sound. In later weeks, oil was difficult to detect in direct measurements, but transplanted clean mussels suspended in cages near contaminated beaches accumulated significant concentrations of oil during 30 day *in situ* tests. Mussels are excellent accumulators of hydrocarbons, have low hydrocarbon metabolizing rates, and indicate that hydrocarbons were available to organisms below the surface. The gas chromatography-mass spectroscopy (GC-MS) profiles of the hydrocarbons in the mussels by Short and Rounds (1993b) indicate that mussels suspended in cages below the surface were exposed to particulate oil, rather than only the lighter fractions with relatively higher water solubility. These tests indicate that petroleum hydrocarbons were biologically available below the surface.

Sediment traps near oiled beaches collected oil-contaminated sediment in

1990 and 1991, indicating that there were physical mechanisms in place to transport low levels of contaminated sediment, probably from intertidal beaches, and deposit them on submerged sediments (Sale et al. 1993). Although the quantities were not acutely toxic, they indicated that oil movement to submerged sediments persisted 1-2 years after the spill.

Hydrocarbon contamination of subtidal sediments was measured inside and outside Prince William Sound (O'Clair et al. 1993) during 1989 and 1990. Subtidal hydrocarbon concentrations were generally less than contaminated beaches, but were readily detected at a variety of sites in Prince William Sound from 0-20 meter depths. Concentrations were usually highest at the shallower depths, and from sites near heavily contaminated beaches, such as Herring Bay. Depths of 40 and 100 meters sometimes had low levels of hydrocarbon contamination, but the analyte profiles were not always similar to *Exxon Valdez* crude oil. By 1990 there was some indication at some of the heavily contaminated sites that hydrocarbon levels at depth increased. This may be the result of redeposition of weathered oil by natural processes or from extensive cleaning activities moving some of the oil down-slope.

Bacteria numbers in subtidal sediments were significantly greater in oiled areas compared to control sites (Braddock et al. 1993), indicating that the hydrocarbon contamination was providing a substrate for bacterial action. These measurements were made on sediments collected along with the sediments measured by O'Clair et al. (1993). There is a high correlation between the hydrocarbon concentration and the bacterial numbers. Increases in bacterial numbers re-

flect one of the first processes instrumental in sediment recovery (bacterial degradation of hydrocarbons), but also provides another route of hydrocarbon entry into higher organisms. Access to the petroleum hydrocarbons can be direct by contact with the sediments, or indirect through the food chain.

The composition and density of the meiofauna (small benthic animals) community in intertidal and subtidal sediments varied between oiled and unoled bays (Shirley et al. 1993). Harpacticoid copepods, which have highest densities in surficial (the upper few millimeters) sediments, generally had lower densities in the intertidal (0 m) zone of oiled bays in initial samples following the spill, but not subtidal (-6 m) sediments. In contrast, the numerically predominate nematodes, which reside deeper in the sediments and are anaerobes, exhibited little effect in abundance.

Field experiments examining meiofauna recolonization in trays of oiled and unoled sediments found that colonization was rapid for many true meiofaunal taxa (but was much slower for the deeper dwelling nematodes), even in highly oiled sediments. However, the abundance of most taxa, including two harpacticoid species and total copepods, was significantly depressed at high oil dosages early in the experiment, but not after 29 days (Fleeger et al. 1993; Shirley et al. 1993). Differences in colonization rates rather than oil-induced mortality was the apparent cause. However, abundances of a few taxa (e.g., ostracods) remained depressed in highly oiled sediments for more than a year. Differences between oiled and unoled bays diminished in years subsequent to the spill.

Several subtidal fish species were studied following the spill. Exposure of

fish using the littoral zone (shore or coastal zone), particularly Dolly Varden, was greater in the months following the spill (Varanasi et al. 1992; Collier 1993). A year later, in 1990, Dolly Varden exposure to oil decreased. However, nearshore benthic species (rock sole, yellow fin sole, and flathead sole) continued to be exposed to hydrocarbons. Pacific halibut occurring in depths greater than 30 meters appear to have less exposure than fish found in shallow subtidal areas. Pollock were found to have increased exposure to hydrocarbons even as far as 400 miles from Bligh Reef (Varanasi et al. 1992; Collier 1993). Pink salmon fry collected from oiled areas in 1989 were contaminated with petroleum, particularly in the viscera, but no hydrocarbons were detected in fry collected from the same areas in 1990 (Wertheimer et al. 1993).

Several subtidal communities were affected by the oil spill. Benthic communities in the fjordic portion of Herring Bay showed signs of impact, including declines in species diversity associated with increasing dominance by a single polychaete (worm) species (Jewett et al. 1993a). Populations of leather stars and helmet crabs were much less abundant in oiled areas as compared to non-oiled areas (Dean and Jewett 1993) and eelgrass had lower densities of turions and flowers at oiled sites (Dean et al. 1993).

Predicting recovery of the subtidal zone is complicated by the habitat and species diversity and the varying responses of plant and animal populations to oiling. Also, since the oil tended to move down slope over time, some subtidal communities were affected later than others. Concentrations of oil in some shallow subtidal sediments decreased over time, but other subtidal sediments showed no significant changes

during the study period (O'Clair et al. 1993). Eelgrass and some species of subtidal algae appeared to be recovering after being affected by the oil. The free space created by the loss of algae was being recolonized by new recruits (Dean et al. 1993b). In some eelgrass beds, sensitive burrowing amphipods recovered to near prespill densities by 1991 (Jewett and Dean 1993b). Leather stars showed little sign of recovery through 1991 (Dean and Jewett 1993). Helmet crab populations appeared to be changing, but it is unclear if population recovery or immigration from non-oiled areas accounts for the increased abundance of this species in oiled areas.

Oil contamination did reach the water column and the shallow subtidal sediments. The lack of other pollutants in Prince William Sound allows for a suite of unique studies to examine the impacts of low level subsurface oil contamination over time. The oil contamination levels were not acutely toxic like the physically smothering oil that impacted some upper intertidal zones, but the low level contamination did cause increases in bacterial numbers, meiofauna mass and species structure, and did alter several subtidal habitats. There are definite signs of recovery in the habitat studies, yet the low level sediment contamination continues to be detected. The residual oil contamination in beaches and under oiled mussel beds may continue to be a reservoir of oil for redistribution to nearby subtidal habitats for some years to come.

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Summary of Injuries to Fish and Shellfish Associated with the Exxon Valdez Oil Spill

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On March 24, 1989, the *Exxon Valdez* oil tanker ran aground on Bligh Reef in Prince William Sound and spilled nearly 11 million gallons of crude oil. In the days and weeks that followed, oil spread across much of Prince William Sound, the waters off the Lower Kenai Peninsula, Afognak Island, Kodiak and the Alaska Peninsula. Birds, otters and seals were obviously in harm's way, but since oil floats, many thought that fish could swim away from the danger above. Not all could.

Rockfish were the only adult fish found dead following the spill (Andrew G. Hoffman, personal communication). Determining the cause of death in a fish is difficult unless it recovered shortly after the animal has died. Nevertheless, five rockfish were found sufficiently fresh to determine oil as the cause of death. Despite this, most rockfish live at depths that oil was not known to have reached in the first few months following the spill. Nevertheless, demersal rockfish in early May 1989 had significantly higher concentrations of hydrocarbons and hydrocarbon metabolites in their bile in oiled than in non-oiled areas. Over time, more of the heavier fractions did reach these depths and rockfish tissues collected in the fall of 1991 (the most recent samples tested) still showed signs of chronic histopathology (Gary D. Marty, personal communication).

Though rockfish were the only adult fish observed dead following the spill, small intertidal and juvenile fish which

may have been killed would not have been noticed in the omnipresent mousse.

It was unfortunate that herring were just beginning their near-shore and intertidal spawning when the oil spill occurred. The oil did not deter them, however, and they spawned on the oiled shores and kelps with their usual abandon. Adults, eggs and juveniles were exposed to oil. The hatching rate was lower, there were more chromosomal aberrations in the larvae and the proportion of viable larvae was lower in the oiled areas (Evelyn D. Biggs, personal communication).

Three years later when the fish in this year class began to mature, they represented the next to smallest recruitment of 3-year olds to the spawning population in 25 years despite that they themselves were the result of a strong year class. Every four to six years, one year class of herring usually recruits to the spawning population at a significantly higher level than other year classes and dominates the spawning population until its numbers decline with time and another large year class takes its place. The 1988 year class was such a class. During the oil spill, the 1988 year class was exposed to oil in its rearing areas. It began to dominate the spawning population in 1992, yet the fertility rate of the eggs it produced was significantly lower in the oiled areas than in the un-oiled areas (Richard M. Kocan, personal communication).

During the time of the oil spill, young

salmon were leaving their natal streams and hatcheries for the open ocean. The oil did not seem to diminish the available food for salmon juveniles in the oiled areas (Alexander C. Wertheimer, personal communication), but the extra metabolic energy expended by juveniles to detoxify the water soluble fractions of oil to which they were exposed may have been the cause of slower growth rate found in oiled areas compared to unoiled parts of Prince William Sound (T. Mark Willette, personal communication). Reduced growth rate, according to Willette, results in poorer juvenile to adult survival. This was observed when the following year pink salmon adults returned at half the rate to a hatchery in an oiled area as to hatcheries in the unoiled parts of Prince William Sound.

In the fall of 1989, pink salmon returned to spawn in the intertidal portions of Prince William Sound streams. Where oil was present in the spawning gravel, eggs and fry suffered higher mortalities than in areas of clean gravel (Samuel Sharr, personal communication). The upper region was the most heavily oiled of the intertidal areas, and it was the slowest to be cleaned by tides and waves and other natural scouring actions. The following year, egg and fry mortalities were significantly higher only in the upper intertidal portions of oiled streams. However, in 1991, mortalities in all intertidal regions were higher in the oiled than in the unoiled areas. The same phenomenon appeared in 1992 as well. It is theorized that genetic damage occurred when the adults spawning in 1991 and 1992 were incubating as eggs and fry in oiled gravel two years earlier. Genetic and environmental causes of this apparent functional sterility are currently being investigated (Samuel Sharr and

James E. Seeb, personal communication).

Oil was still present in the salmon fishing areas when adult salmon returned in the summer of 1989. Nets could not avoid straining oil and water; oiled nets contaminated the salmon held by them; and oil-tainted salmon could not be sold. Fishing seasons were closed and many more adults returned to some spawning streams than was desired. It appears that the excess sockeye salmon returning to Red Lake on Kodiak Island and to the Kenai River system on the Kenai Peninsula produced more juvenile salmon than the ecosystems' food webs could support (Kenneth E. Tarbox, personal communication, Dana C. Schmidt, personal communication). Apparently, many young fish starved, fewer than normal outmigrated in the following years as smolts and fewer than normal are expected to return as adults; so few in fact, that commercial and sports fishing seasons may be closed. If this happens, hundreds of millions of dollars will be lost from the commercial and sports fishing industries.

In the Kenai system, the effects of other overescapements in the two years prior to the *Exxon Valdez* oil spill combined with the effect of the 1989 oil spill to severely impact that river system. There has been no indication of recovery to date.

Dolly Varden and cutthroat trout were overwintering in freshwater lakes when the spill occurred in Prince William Sound, but they soon left these lakes to forage in the near shore areas until they once again entered freshwater in the fall. The rocks and sediments of the near-shore areas were coated with oil and long after oil had left the pelagic waters, the near-shore was still contaminated. Some of those areas which were

cleaned by response crews were devoid of life because of the cleaning process. Dolly Varden and cutthroat trout frequenting these areas may have found less food in the cleaned areas and toxic hydrocarbons in the oil-contaminated locations. Subsequent sampling found their growth rates and annual survival were less in the oiled than in the unoiled areas (Kelly R. Hepler, personal communication). Some populations of cutthroat have declined to such critically low levels that these areas are now closed to fishing.

Clams were impacted by some of the methods used to clean the beaches following the oil spill. Many clams on oiled but uncleaned beaches survived, but their growth rates appear to be lower than in the unoiled areas (J.D. Johnson, personal communication).

Oil is known to have a very severe impact on crustaceans, but commercial fishing and heavy predation by an expanding sea otter population prior to the *Exxon Valdez* oil spill made it very difficult or impossible to determine the effect of oil on some species. A dungeness crab project quickly came to an end when only one crab could be found in the impacted area of Prince William Sound (Charles Trowbridge, personal commu-

nication). The Green Island area was directly in the path of much of the oil passing through Prince William Sound and it had once been a very productive area for commercial shrimp fishing. But the population crashed before the spill and therefore determining injury due to oil is very difficult. Nevertheless, in the absence of commercial fishing over the last several years, this population has not recovered (Trowbridge, personal communication). As noted earlier, recent evidence suggests that rockfish continue to be exposed to oil. It logically follows that shrimp in the same habitats would also be exposed, but whether this is preventing these populations from recovering is unknown.

The fish and shellfish of the spill areas were impacted by the oil even though they were beneath the surface. Because many of the fish and shellfish examined are commercially important species, it has often been difficult to separate the effects of oil from fishing mortality. Nevertheless, within sometimes broad boundaries, it has been shown that even the adult populations of fish and shellfish were affected by impacts even to the juvenile stages of these animals. Restoration considerations are warranted and may be necessary in order to bring some of these stocks back to healthy levels.



How Do You Fix the Loss of Half a Million Birds?

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The *Exxon Valdez* Oil Spill occurred at the end of March 1989, near the end of winter when thousands of Bald Eagles and loons, plus countless waterfowl and seabirds were overwintering in Prince William Sound. The oil spread slowly at first, then high winds blew the oil quickly southwest across the Sound into sheltered bays and fjords densely populated by sea otters, harbor seals and birds. The oil filled bays acted as a trap for birds flying into the sheltered coves to roost for the night. Western Prince William Sound became a dead zone overnight.

The wind continued blowing the oil out of the Sound into the Gulf of Alaska and along the Kenai Coast fouling the headlands and disrupting the birds at the Chiswell Islands, but luckily causing relatively little injury to the seabird populations there.

In early April oil reached the Barren Islands just prior to the beginning of nesting by Common Murres on the seabird breeding cliffs. Many tens of thousands of breeding Common Murres had gathered in large rafts adjacent to the islands and the oil swept away the great majority of these birds. Prior to the spill the breeding colonies on the Barren Islands were home to approximately 130,000 birds, but 60-80 percent were engulfed, carried away and killed by the oil. Many of the carcasses drifted ashore on the Alaska Peninsula in April and May, and many carcasses undoubtedly drifted into the Gulf of Alaska and sank without ever reaching shore.

The timing of the oil surrounding the

Barrens could not have been more devastating to the murres, but it fortunately did not kill any puffins or storm petrels which also breed there in large numbers. Luckily, storm petrels and puffins normally do not return to the breeding colonies until mid May, and most of the oil had drifted away from the Barren Islands before these birds arrived at the colony. Almost certainly, some individuals of these species were killed at sea in the Gulf of Alaska and Shelikof Straits, with little likelihood of the birds being recovered on shore.

The oil continued past the Barrens, both eddying into Cook Inlet and Kachemak Bay, and continuing south into the Gulf of Alaska and Shelikof Strait, oiling beaches on the east and west sides of Afognak and Kodiak Islands.

During the initial response phase about 36,000 birds were recovered dead from beaches, and an additional 1604 live, oiled birds were taken to rescue centers in Valdez, Seward, Homer and Kodiak for cleaning and rehabilitation. Eight hundred and one of these birds were successfully cleaned and released back to the wild. Their survival was not assessed.

Some of the 36,000 birds found dead on beaches in the late summer of 1989 undoubtedly were natural mortality not associated with the oil spill, but only a very small proportion of the oiled birds were ever recovered from the more than 1,300 miles of oiled beaches. Several factors contributed to the small percentage carcass recovery. Many birds were

killed, became waterlogged and sank before reaching shore, and some may have been washed from shore along with oil during the tidal cycles. Other birds were scavenged by eagles, ravens and gulls or by foxes and bears attracted to the beaches because of the carcasses present. Radio marked carcasses put out as part of the Damage Assessment remained on beaches only an average of 24 hours, with most of the carcasses being carried inland by scavengers patrolling the beaches. Many carcasses may have been buried in the beaches by wave action, and many miles of beach were searched only rarely by personnel retrieving carcasses. Small birds were probably missed, and many oiled birds still alive when they reached shore may have lurched up the beach and hidden in the undergrowth to avoid detection by beach walkers, cleanup workers or predators. The Trustees' extrapolation of the number of birds oiled in the spill is necessarily uncertain, but indicates very strongly that 350,000 to 500,000 birds were trapped and carried by the advancing oil as it swept out of Prince William Sound across the Gulf of Alaska and against the Alaska Peninsula. How many additional birds were carried into the Gulf of Alaska is unknown.

Because Prince William Sound is an overwintering area for birds which migrate away from the spill zone to breed elsewhere, it was not possible in 1989 to accurately assess the numbers of birds present during the spill event. Loons, several species of seaducks, many tens of thousands of migrating shorebirds and thousands of Bald Eagles were all present during the initial time of the spill, but most left the spill area to breed elsewhere. The number of migrants killed and the proportion of these birds that

were lightly oiled and suffered reproductive problems is unknown. The spill area was too large to be able to accurately assess the number of birds at risk.

Birds staying in the spill zone remained at risk of exposure for a very long time. Hundreds of miles of beaches became oiled, and a substantial portion of the oil remained aground, but much oil bled from the beaches with each high tide cycle. The ten to 15 foot tidal excursion in Prince William Sound insured maximal intertidal exposure and penetration of oil into the coarse cobblestone beaches throughout much of the spill area. While most of the oil washed ashore or sank, some oil continually bled from beaches for months posing a risk to birds on the water. More insidious, however, was oil contaminating intertidal invertebrates and fish eggs needed as food by migratory and breeding birds. The persistence of oil in mussel beds and intertidal seaweed beds in some places has lasted for years, with continued exposure to species such as Harlequin Ducks, oystercatchers, and some gulls such as Black-legged Kittiwakes and Mew Gulls. In 1989, kittiwakes used seaweed (primarily *Fucus*) to build their nests and the oil contaminated eggs. Reproductive failure of some kittiwake colonies may have been related to oil exposure.

The extensive cleanup activities in 1989 (and to a lesser extent in 1990) caused disturbance of potential breeding birds in addition to the mortality of directly oiled birds. Bald Eagles, Marbled Murrelets, Pigeon Guillemots, Harlequin Ducks and Black Oystercatchers all suffered breeding losses as a result of oiling, disturbance, or a combination of the two in 1989 and 1990.

Over the past 3 years, the injury suffered by most species appears to have

ended with the cleanup of oil, and many species have apparently returned to normal breeding, but by a reduced population. Marbled Murrelets and Bald Eagles appear to have returned to normal breeding, but Black-legged Kittiwakes have not been carefully monitored since 1989.

Some species, however, are still suffering continued injury. Black Oystercatchers, Harlequin Ducks and Pigeon Guillemots are heavily dependent upon the nearshore and intertidal environments and showed injury in 1990 and 1991. Results of the 1992 breeding season were not available for all species at the time of this writing, but Harlequin Ducks certainly showed continuing injury.

The largest scale continuing injury to reproduction in birds is with Common Murres. These seabirds breed in very dense colonies in only a few locations, and the large colonies at the Barren Islands, and the Alaska Peninsula (Puale Bay and Ugaiushak Island) suffered such great mortality in 1989 that their social structure and organization remain severely disrupted. Many young birds, apparently attempting to breed for the first time at age 4 or 5 years, have returned, but the courtship and egg laying patterns of the birds are poorly synchronized and occur nearly a month later than they should each year. The fragmented, late breeding has resulted in increased predation of eggs and chicks by gulls and ravens, and the winter storms have swept more than 100,000 young chicks off the cliffs to their deaths because the breeding has been so late in 1990, 1991 and 1992. The continuing abnormal breeding at these colonies is very disturbing, as the murres appear to be in real danger of becoming permanently entrained to late breeding, possi-

bly because young birds have established the wrong patterns. If this is permanent, the prospects for these colonies is poor, because a breeding failure will lead to the eventual decline and extinction of these colonies. Many hundreds of thousands of additional Common Murres will be lost if this injury continues. Some bold restoration efforts must be attempted to try to reverse the trend with Common Murres.

Restoration opportunities by the Trustees have been made possible by the unprecedented settlement with Exxon after the spill. The 900 million dollars should provide funding for many diverse projects which have been proposed. The Trustees now have the opportunity to try many innovative new techniques to assist the species showing severe or continuing injury.

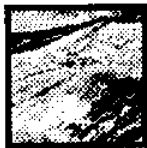
Much of the pristine old growth forest of Prince William Sound, the Kenai Peninsula and Afognak Island is privately owned and available for logging, with potential to harm breeding Marbled Murrelets, Harlequin Ducks and Bald Eagles. Protection of these breeding habitats is important, and a substantial proportion of settlement monies should be used to acquire valuable forest parcels in imminent threat of logging.

More important, however, is the need to help restore normal breeding to Harlequin Ducks and Black Oystercatchers and to assist Common Murre and Black-legged Kittiwake colonies suffering breeding failure. Careful cleaning and restoration of specific intertidal feeding habitats, including restoration of mussel beds and seaweed beds is essential. Without this intertidal food base, recovery of the birds and sea otters dependent upon them will not occur normally. The murre and kittiwake colonies remain the most

difficult to help. Control of predators, both foxes and some bird species, should help prevent further losses, but active measures are also needed for murre. Since the breeding synchrony and timing are still wrong at some colonies, drastic measures are called for to assist these fragile colonies. Restoration of colonies of closely related birds has been accomplished in Maine and the Galapagos by placing decoys and playing recordings of courtship calls over loudspeakers to stimulate the birds to begin breeding at

the correct time. These techniques should be tried as pilot projects in Alaska, and if successful, a large program to restore the murre colonies should be attempted.

The *Exxon Valdez* oil spill injury and settlement were both unprecedented in U.S. history. The recovery efforts should also be bold, and creative in the pioneering spirit of Alaska, especially since Exxon has provided sufficient funds to allow the Trustees the liberty to use many different techniques and projects to help restore this unique place.



Effects of the *Exxon Valdez* Oil Spill on Marine Mammals in Prince William Sound

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Prince William Sound is a large protected embayment that provides excellent and diverse habitat for a variety of marine mammal species. The most common of these species are sea otters, harbor seals, killer whales, humpback whales, Steller sea lions and harbor and Dall's porpoises. When the *T/V Exxon Valdez* spilled 11 million gallons of crude oil in Prince William Sound on 24 March 1989, many of these marine mammals swam or crawled through oil and inhaled aromatic hydrocarbons as they breathed at the air/water interface. Harbor seals and sea otters rested on oiled rocks and algae. Seal and sea otter pups were born on oiled substrate and nursed on oiled mothers. The prey of some species were contaminated by oil, and this food chain contamination may have lasted for some time after the spill.

Shortly after the spill occurred, studies were initiated to assess its effects on marine mammals that were likely to be impacted. For some species, baseline data on abundance, seasonal distribution, natural mortality, and reproduction were so incomplete or lacking that it was difficult to conduct meaningful studies (e.g., harbor and Dall's porpoises). Others, such as sea lions, were present for only a short period after the *Exxon Valdez* spill and then moved away as part of their annual cycle, and it was not possible to track which animals had been in

contact with the spill. For all species, an undetermined but significant amount of time is spent underwater, making them difficult to count and observe. For some, ongoing declines (harbor seals) or fishery interactions (killer whales) made interpretation of results difficult. Because of the inadequacies of comparative data and study methodology, many of the effects of the *Exxon Valdez* oil spill may never be known definitively. Findings of the major studies are presented here.

Humpback whales are present and feed in Prince William Sound from spring through autumn. In order to assess the impacts of the spill on humpbacks, photographs of individual whales in Prince William Sound were collected from May to September 1989-1990. Research vessels traversed over 20,000 nm in search of whales during the two field seasons. Photographic analysis of Prince William Sound humpbacks revealed 59 identifiable whales in 119 encounters in 1989 and 66 whales in 201 encounters in 1990. No measurable decline in the number of humpbacks occupying Prince William Sound occurred following the *Exxon Valdez* oil spill. Fewer humpbacks used the Lower Knight Island Passage area in 1989 than in 1988 or 1990. Increased vessel and aircraft traffic associated with cleanup activities in that area may have caused redistribution to other areas. No observations were made of humpbacks

swimming through oil. There were no reports of dead, stranded humpbacks during the 1989 or 1990 field seasons.

Approximately 245 "resident" and 52 "transient" killer whales were known to use Prince William Sound prior to the oil spill. In order to assess the impact of the *Exxon Valdez* oil spill on killer whales, a photographic identification study of individual killer whales was conducted in Prince William Sound during 1989-1992. During the study, a maximum of nine resident pods (148 whales) and four transient pods (34 whales) was documented. Following the *Exxon Valdez* oil spill, five different pods of killer whales were seen swimming through oil, and killer whales were also observed rubbing on an oiled beach. Analysis of photographs of resident pods in 1989 revealed two animals missing from AE pod, 22 missing from AN pod, and seven missing from AB pod. Losses from AE pod were within the expected mortality rate. The 22 animals missing from AN pod may have belonged to a subgroup that travelled separately from the main pod in 1989; the missing animals were seen again in 1990. During 1990-1991, no additional animals were documented to be missing from resident pods other than AB pod, which lost an additional seven animals. However, in 1990-1991, ten (and possibly eleven) killer whales were missing from transient pod AT. Three of these missing animals were photographed very close to the *Exxon Valdez* on or about 27 March 1989.

The loss of animals from AB pod in 1989-1991 is unusual and higher than normal mortality would explain. These losses translate to mortality rates of 19.4% in 1989-1989 and 20.7% in 1989-1990, compared to 3.1% in 1988, 4.3 % in 1991, and 0 in 1992. No calves were observed in AB

pod in either 1989 or 1990. A single calf was seen in 1991 and two calves in 1992. All other resident pods in Prince William Sound have maintained or increased their numbers since 1988.

Harbor seals occur in Prince William Sound throughout the year, particularly in the coastal zone, where they feed and haulout to rest, pup and molt. Some of the largest haulouts in the Sound, and waters adjacent to those haulouts, were directly impacted by oil. In oiled areas of Prince William Sound, 50%-100% of the seals and their pups became oiled. Seals did not appear to avoid oil in the water or on haulouts. Oiled seals in oiled areas were reported to be sick, lethargic, or unusually tame.

Microscopic examination of tissues from seals collected following the *Exxon Valdez* oil spill found debilitating lesions in the brains of oiled seals. Exposure to aromatic hydrocarbons caused swelling and degeneration of nerve axons, which could have made it very difficult for seals to perform normal tasks such as swimming, diving, and feeding. Hydrocarbon metabolites in bile were 7-13 times higher in seals collected from oiled parts of Prince William Sound than in those from the Gulf of Alaska. This confirms that seals took oil into their bodies through contact, inhalation, and/or ingestion. Because seals have enzyme systems that allow them to detoxify and excrete hydrocarbons, the levels found in most tissues were very low. Highest levels occurred in blubber and in milk.

Aerial surveys conducted in 1989-1992 showed that pup production was lower in oiled areas during 1989 than in 1990-1992, while in unoiled areas it was the same. Based on aerial surveys and carcass counts, neonatal pup mortality was estimated to be over 20% in some

oiled areas. Prior to the *Exxon Valdez* oil spill, there was an ongoing decline in the number of seals in Prince William Sound that was similar at oiled and unoiled sites. After the *Exxon Valdez* spill, counts during the autumn molt showed a much greater decline at oiled sites (45% compared to 8% at unoiled sites). In 1992, there were still 35% fewer seals at oiled trend count sites than in 1988, compared to 18% fewer at unoiled sites, indicating that recovery has not occurred.

The number of dead seals found and reported greatly underestimates the number that died, since dead seals do not float but sink to the bottom. Therefore, the number of seals that died as a result of the *Exxon Valdez* oil spill was estimated based on aerial surveys conducted during the molt.

Based on the assumption that the trend in numbers at unoiled sites was "normal" and that any greater decline at the oiled sites was due to the oil spill, calculations indicate that there were approximately 350 fewer seals in Prince William Sound than would have been expected if the *Exxon Valdez* spill had not occurred.

Sea otters are widespread throughout Prince William Sound and they were severely damaged as a result of the *Exxon Valdez* oil spill. They rely on their fur to keep them warm, and consequently they were particularly vulnerable to oiling which destroyed its insulative value. They ingested large amounts of oil as they attempted to clean their fur by

grooming. Carcasses of 781 sea otters that were judged to have died because of the *Exxon Valdez* spill were recovered in or adjacent to the oil spill area, with 424 of those from Prince William Sound. An additional 123 otters died at rehabilitation centers. More sea otters were undoubtedly oiled and died, but their carcasses were not recovered. It is estimated that total sea otter mortality in Prince William Sound was over 2,000, and that in the entire spill area mortality may have exceeded 4,000. Necropsies of otters that died following the spill indicated a high incidence of pulmonary emphysema and gastric erosion and hemorrhage, in addition to kidney and liver damage.

An unusually high proportion of prime-age adults died in both the spill year and in post-spill years relative to pre-spill years. Before the spill, mortality was highest in juveniles (45%) and aged (40%) individuals, and relatively low (15%) in prime-age adults. Following the spill and continuing through 1991, mortality of prime-age adults increased to 43-44%. This suggests that there are prolonged, spill-related effects on the sea otter population as a result of the *Exxon Valdez* oil spill.

Pre-spill and post-spill boat-based surveys indicated that between 1984-1985 and 1989-1990, sea otter abundance increased 13% in unoiled areas compared to a 35% decrease in oiled areas. By 1991, otter abundance in oiled areas had apparently stabilized, but was still below pre-spill levels.

Restoration Planning Following the Exxon Valdez Oil Spill

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The restoration planning process following the Exxon Valdez oil spill has focused on identifying, evaluating, and integrating information about the nature, extent, and persistence of injuries to natural resources and services, the rate and adequacy of natural recovery, and the opportunities for restoration. This process changes as new information is received, but will culminate in the publication of a Draft Restoration Plan in Spring 1993. The damage assessment and restoration science studies are the primary sources of information on injuries; other sources include data collected during the oil spill cleanup, public comments, and the scientific literature.

This paper reviews the planning approach taken by the Restoration Planning Work Group on behalf of the Exxon Valdez Oil Spill Trustees to identify and evaluate restoration options for inclusion in the draft restoration plan. I also provide a brief vision of how the plan may be implemented, once public comment has been considered.

Identification of Restoration Options

The restoration planning process has identified the widest possible range of restoration ideas, based on suggestions from a public symposium (RPWG, 1990a), public "scoping" meetings (RPWG, 1990b), and a technical workshop (unpublished). Restoration ideas have been organized into restoration options and databases necessary for their evaluation have been assembled. Thirty-five restoration options were identified and pre-

sented to the public in the Exxon Valdez Oil Spill Restoration, Volume 1 - Restoration Framework (Exxon Valdez Oil Spill Trustees, 1992a) for review and comment. Based on public comment as well as additional input from the Trustee Agencies and independent scientific peers, there are presently 32 restoration options being considered for inclusion in the draft restoration plan. Arranged by category of restoration activity, they are:

Management of Human Uses

1. protect archaeological resources,
2. intensify management of fish and shellfish,
3. increase management for fish and shellfish that did not previously require intensive management,
4. reduce disturbance at marine bird colonies and marine haul-out sites and rubbing beaches,
5. reduce harvest by redirecting sport-fishing pressure,
6. create national recreation area or wilderness area,
7. increase management in parks and refuges,
8. seek voluntary restrictions in subsistence harvests of marine and terrestrial mammals and sea ducks,
9. seek restrictions and closures to legal harvests of terrestrial mammals and sea ducks,
10. minimize incidental take of marine birds by commercial fisheries,

Manipulation of Resources

11. preserve archaeological sites and ar-

- tifacts,
12. improve freshwater wild salmonid spawning and rearing habitats,
 13. create new recreation facilities,
 14. eliminate residual oil from important intertidal habitats, e.g., mussel beds,
 15. accelerate recovery of upper intertidal (*Fucus*) zone,
 16. supplement subtidal substrates (algal and other) for spawning herring,
 17. test feasibility of enhancing murre productivity,
 18. eliminate introduced foxes and other predators from islands important to nesting of marine birds,
 19. replace fisheries harvest opportunities by establishing alternate salmon runs,

Habitat Protection and Acquisition

20. update and expand the State's Anadromous Fish Stream Catalog,
21. acquire tidelands,
22. designate protected marine areas,
23. acquire additional marine bird habitats
24. acquire "inholdings" within parks and refuges,
25. protect and acquire upland forests and watersheds,
26. acquire extended buffer strips adjacent to anadromous fish streams,
27. designate and protect "benchmark" monitoring sites,
28. acquire access to sport fishing streams,
29. establish or extend buffer zones for nesting birds,

Other Options

30. test subsistence foods for hydrocarbon contamination,
31. establish a marine environmental institute, and
32. replace (return) archaeological artifacts.

Development of Injury Criteria and Identification of Resources and Services that Warrant Restoration

To decide whether it was appropriate to spend restoration funds on a particular resource or service, criteria were first developed that evaluated available evidence for consequential injury and the adequacy of natural recovery. "Consequential injury" indicates a loss attributable to exposure to *Exxon Valdez* oil, or otherwise attributable to the oil spill and cleanup. "Loss" for injured natural resources is defined as:

1) significant direct mortality, 2) significant declines in population size or productivity, 3) significant chronic and sublethal effects, or degradation of habitat due to contamination by oil or cleanup.

A natural-resource service has experienced "consequential injury" if the oil spill or associated cleanup has: 1) significantly reduced the physical or biological functions performed by the natural resource, 2) significantly reduced aesthetic, intrinsic, or other indirect uses provided by the natural resources; or in combination with either of these, 3) resulted in the continued presence of oil on lands integral to the use of special purpose lands (i.e., parks and refuges designated by the State of Alaska or Federal Government for the protection and conservation of natural resources and services)

To maximize the benefits of restoration expenditures, the Trustees may consider the effects of natural recovery before investing restoration dollars. In a scientific sense, full ecological recovery has occurred when the flora and fauna are again present in similar numbers, health, and productivity to pre-spill conditions, and there is a full complement of age classes. A fully recovered ecosystem is one which provides the same func-

tions and services as were provided by the pre-spill, uninjured system.

For each injured resource and service, an estimation of the rate of recovery will be made based on the best information available from the damage assessment and restoration-science studies, the scientific literature, and other sources. If it appears that recovery will be nearly complete before the benefits of a restoration study or project can be realized, then the Trustees may determine that spending funds is not justified. However, if it appears that the recovery time will be prolonged, it may be worth implementing technically feasible, cost-effective restoration options.

Criteria to Evaluate Restoration Options

To help determine which of the many restoration options are most appropriate and beneficial, the following criteria were developed based largely on the Comprehensive Environmental Response, Compensation and Liability Act of 1980 (42 U.S.C. 9601):

1. potential to improve the rate or degree of recovery,
2. potential to prevent further degradation or decline,
3. technical feasibility,
4. degree to which proposed action benefits more than one resource or service,
5. degree to which proposed action enhances the resource or service,
6. potential for additional injury to either resources or services from implementation of option, and
7. the relationship of expected costs to expected benefits.

These criteria have been used throughout the planning process. All ideas developed from the initial public "scoping" meetings also were screened

against these criteria during a preliminary evaluation. Ideas which were not technically feasible, or which could produce significant additional injury upon implementation, were rejected.

Evaluation of Restoration Options for Identifying and Protecting Marine and Upland Habitats

Additional steps will be needed to properly evaluate habitat protection and acquisition options. While a final process has not been adopted, the Trustees issued a Restoration Framework Supplement (*Exxon Valdez Oil Spill Trustees*, 1992b) that proposed a detailed habitat protection process for public review and comment. The steps in this process include:

1. identification of key upland habitats that scientific data or other relevant information link to the recovery of injured resources and services. This includes an analysis of imminent threat from development (e.g., logging or mining), that recognizes the need to respond to a proposed change in land use that could foreclose habitat protection or other restoration opportunities.
2. characterization and evaluation of potential impacts from changed land use relative to their effects on recovery of the injured ecosystem and its components; comparative evaluation of recovery strategies not involving acquisition of property rights (e.g., redesignation of land-use classification), including an assessment of protection afforded by existing laws, regulations, or other alternatives.
3. evaluation of cost-effective strategies to achieve restoration objectives for key upland habitats identified through steps one and two above.

Restoration alternatives for resource injuries would be evaluated.

- 4) willing seller-buyer negotiations with private landowners for property rights, and
- 5) public management of acquired property rights.

Development of Restoration Alternatives

The draft Restoration Plan will describe a reasonable range of restoration alternatives based on requirements of the National Environmental Policy Act of 1969 (40 CFR 1500-1508). The consequences and impacts of each alternative must be analyzed in an Environmental Impact Statement (EIS) (Council on Environmental Quality, 1986). A programmatic EIS will be published simultaneously with the restoration plan.

Each alternative will consist of several or more (a set) of the restoration options listed above. More than one restoration option can be used to restore any one injured resource or service. One option also could address the restoration of multiple injured resources and services. Each alternative, then, will achieve restoration through a different set of options.

We do not implement all the restoration options listed above because their combined cost would greatly exceed the funds now available. As a consequence, alternatives consisting of different sets of restoration options are constructed for public review. In this way public preferences on the options are collected and the implications of choosing some options over others become evident. After consideration of public comment, the Trustees will choose one or more of the alternatives for implementation.

Six possible restoration alternatives were identified in the Restoration Frame-

work (*Exxon Valdez Oil Spill Trustees, 1992*), they are presented here for discussion only and do not at this time indicate any preference of the Trustees. They are:

1. *No action* - This alternative is to undertake no active restoration but to rely on natural recovery to restore the injured ecosystem and its associated services.
2. *Management of human uses* - The use of existing State of Alaska and Federal Management authorities to modify human uses of injured resources or services is emphasized in this alternative.
3. *Manipulation of resources or services* - This alternative focuses on measures taken directly (usually on-site) to rehabilitate or replace an injured species, restore a damaged habitat or enhance services provided by a damaged resource.
4. *Habitat Protection and Acquisition* - This alternative includes changes in management practices on private and public lands and the creation of "protected" areas on existing public lands and on marine waters to prevent further damage to injured resources. Beyond land management practices, damaged habitats or property rights can be acquired short of fee simple title, e.g., purchase of timber rights.
5. *Acquisition of equivalent resources* - Acquisition of equivalent resources means to compensate for an injured resource by substituting another resource that provides the same or a substantially similar service as the injured resource or service. However, direct restoration approaches (manipulation of resources and services, and habitat protection and acquisition) also can be implemented on an equivalent-resource basis.

6. *Combination alternatives* - Each alternative above may be considered by itself or mixed in a number of ways, depending on priorities and approach. Differences among combination alternatives could be based on the severity of injury, the level (certainty) of knowledge on recovery, the perceived effectiveness of restoration techniques, or where restoration will be implemented. For example, one combination alternative could address restoration of only the most severely injured resources and services that we know are not recovering within the affected area, and that only the most effective direct restoration measures would be used. Another alternative could be less restrictive and address restoration of all injured resources and services, both inside and outside the affected area, and apply all available restoration measures (direct restoration, replacement, acquisition of equivalents, enhancement).

Monitoring

Implicit in each alternative is the provision to monitor the recovery of injured resources and services. It would be the objective of this program element to monitor natural recovery as well as recovery aided by restoration. Monitoring also would be designed to detect latent injuries and reveal long-term trends in the health of ecosystems affected by the oil spill. The duration of the monitoring program would depend on the severity of the effects resulting from the spill and the time necessary to establish a trend for recovery.

Implementation of Plan

Once the public has commented on the Draft Restoration Plan, the Trustees

will select the alternative or alternatives that will constitute the (Final) Restoration Plan. This document is scheduled for publication in Summer 1993. Restoration at the project level will be consistent with restoration options contained within the selected alternative(s) and will begin with implementation of annual work plans beginning in 1994. Each year there will be a call for ideas (project descriptions) for the next years' annual work plan, as there was in 1992 in anticipation of the 1993 field season. Based on this input, a draft annual work plan will be assembled by the Trustees and circulated for public review and comment. After consideration of public comment and any necessary revision, the annual work plan will be adopted and implemented.

Funding

Funding for restoration will come from the \$900 million that the Exxon Companies agreed to pay the United States and the State of Alaska over a period of 10 years. The Exxon Valdez oil spill, however, resulted in injury to resources that may not recover for generations. The extent of injury and the rate of recovery for some resources and services will not be completely known for decades, well beyond the life of the existing settlement. For these and other reasons, restoration needs will continue well beyond the last scheduled payment in 2001. To address this need, the Trustees are considering a proposal to establish an endowment. An endowment could serve to extend the life of the restoration program providing longer-term (perpetual) support for certain restoration activities, e.g., monitoring and research programs, visitors center, and habitat acquisition. An endowment also offers an opportu-

nity to undertake restoration at a different (slower) pace than would be the case if all funds had to be expended within the 10-year life of the settlement. We may not know if initial restoration is successful for many years which suggests a more cautious approach.

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Tracking Exxon Valdez Oil from Beach to Deep-Water Sediments of Prince William Sound, Alaska

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Prince William Sound, a large, complex-fjord-type estuarine system, owes its configuration to plate tectonics and multiple episodes of glaciation. Gravel beaches dominate exposed segments of bedrock islands whose steep submarine slopes are covered with gravelly to muddy sediment. Submerged morainal ridges consist of relict diamicts, indicating rapid tidal currents that sweep away modern fine sediment. Between bedrock and morainal highs of the fjord floor, numerous deep basins (to water depths of 800 m) contain Holocene diatom-rich mud up to 200 m thick. Minor amounts of coarse sediment are being introduced into the estuary from a few fjord-head deltas and as bottom load from the Copper River Delta through Hinchinbrook Entrance. The dominant sedimentary material accumulating today in this fjord complex is fine suspended sediment which is being deposited in deep sediment sinks throughout the sound at rates that vary from 0.3-0.4 cm/yr (Bothner et al. 1990). Much of the insular slope and the fjord walls are kept bare of fine sediment settling from the water column by complex current circulation within the estuary.

We have undertaken four sediment sampling cruises since the Exxon Valdez oil spill in March 1989. The cruises (May 1989, May 1990, August 1990, and June 1992) were planned to sample the bottom sediment along the spill trajectory to follow the geological fate of the spilled oil. Sediment samples were analyzed for

aliphatic and aromatic hydrocarbons.

In May 1989, we sampled bottom sediment at 20 stations along the oil spill trajectory from Bligh Reef southwest through Prince William Sound (15 sites) and along the Kenai Peninsula (5 sites). Each site was chosen after a 3.5 kHz acoustic profile line was run across the prospective sample area (Carlson and Reimnitz, 1990). Most acoustic profile lines over box-core sample sites showed relatively thick accumulations of post-glacial, unconsolidated mud which is accumulating in the deep basins today. Some sites were selected on seafloor highs, and cores from these sites contained pebbly sandy muds, principally a glacial moraine substrate. Oil contamination could not be positively identified in sediment at any of the 15 deep-water sites sampled two months after the spill. Only at one site, near the south end of Prince William Sound, northeast of Latouche Island, did the sediment extract have chemical indicators of possible oil contamination, but the presence of spilled oil could not be verified (Rapp et al., 1990).

Nine deep-water sites, originally sampled in 1989, were occupied 14 months after the spill in May 1990. No visible signs of oil were present in the deep-water sediments, but a consistent increase in the terpane ratios C23/C30 may be a consequence of oil-spill contamination (Kvenvolden, et al., 1991). Relative changes in deep-water samples from 1989 to 1990 in the pristane/phy-

tane ratios suggest changes in the depositional environment which may or may not be due to the effects of the oil spill.

Seventeen months after the spill (August 1990), six islands (Elrington, Knight, Eleanor, Smith, Naked, and Storey) and their insular slopes were investigated (Carlson et al., 1991). We found some oil on all of the beaches we visited. The oil was in a variety of forms including sheens of oil on water that percolated from the beach sediment; thin coatings of oil on sediment or rocks; brown sticky mousse-like patches on sediment and driftwood; and tar or asphalt-like pavements or patches on rocks. In the case of the tar, two chemically similar samples were found about 100 km apart on the beaches of the north side of Storey Island, and on the northwest side of Elrington Island. These tars were not from the *Exxon Valdez* spill because they had different aliphatic biomarker and aromatic hydrocarbon distributions and markedly heavier carbon-isotopic compositions than either the spill oil or the oiled sediments collected from the beaches visited in August 1990. This tar exhibits the characteristics of oil from the Monterey Formation in California (Kvenvolden et al., in press). The other oil samples have sterane and hopane biomarker distributions similar to those of a spilled North Slope crude oil sample secured from the tanker; however, the oils from the beaches are at various stages of alteration as evidenced by hydrocarbon distributions. For example, alkylated naphthalenes and phenanthrenes, as well as *n*-alkanes and isoprenoid hydrocarbons, have partly or completely disappeared (Kvenvolden, et al., 1991).

After sampling the island beaches, we ran high-resolution acoustic profiles

across the adjacent insular slopes and then collected bottom sediment at sites selected from the profiles. Rapid degradation of *n*-alkanes and isoprenoid hydrocarbons limits their usefulness for tracking oil in shallow or deep water sediment. Some of the biomarker characteristics, such as tricyclic-tetracyclic terpane-triplet patterns and sterane/diasterane distributions, suggest addition of spilled oil to sediment at eleven of the shallow water stations occupied in 1990 (Kvenvolden, et al. 1991). All of these samples are located off beaches that were heavily impacted by North Slope crude oil spilled from the *Exxon Valdez*. However, none of the shallow water samples contained visible traces of the spilled oil.

Thirty eight months after the oil spill (June 1992), we found oil from the 1989 spill on beaches of Naked, Green, Knight, Evans, and Latouche Islands, as well as tar from other sources. On the same cruise, five deep water sites previously occupied in both 1989 and 1990, were reoccupied. Samples from these beach and deep water stations are currently under investigation.

Prince William Sound circulation is strongly influenced by the Alaska Coastal Current. This current is affected by fresh water discharge which, according to Royer and others (1990), was at a record low in March 1989, the time of the spill. They concluded that the spilled oil advanced through the sound more slowly than it would have in a normal year. Under these conditions of lower discharge, the amount of suspended sediment carried by streams draining the large glaciers bordering the Gulf of Alaska was probably below normal. Floating oil, even after losing volatiles, is

not dense enough to sink, unless bonded to particulate matter. If the amount of particulate matter is low, the probability of bonding decreases. This process might explain the absence of the spill oil in the deep-water samples collected on the 1989 cruise. On the other hand, if the lower fresh-water discharge caused the "flow through" to be slowed, the oil would have more time to attach to sedimentary particles. However, the general absence of oil in the deep sediment sinks two months after the spill suggests that the first scenario is more likely. By the second summer after the spill, there is evidence that traces of oil are migrating from the oil-impacted beaches down the insular slopes, and meager evidence that deep basin sediment is showing some trace amounts of oil contamination. The samples collected in June 1992 should show whether or not hydrocarbon contamination from the March 1989 oil spill has reached the deep-water sediment sinks of the fjord.

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Characterization of Residual Oil in Prince William Sound, Alaska—3.5 Years Later

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Three and a half years after the T/V Exxon Valdez spill, petroleum still persists on many of the beaches used as study sites in Prince William Sound. Presented are the preliminary results of chemical weathering of residual North Slope crude in an intertidal subarctic environment as viewed from an integrated chemical weathering and physical transport prospective. Residual oil samples collected on various beaches during August 1992 are compared to the original T/V Exxon Valdez cargo oil.

Weathering studies of oil degradation indicate the most resistant components often associated with long-term chronic toxicity are the polycyclic aromatic hydrocarbons (PAHs). These PAH compounds are used as weathering indicators when the normal hydrocarbons are weathered beyond detection. The collection locations were part of a long-term study with the objective of having a close corroboration between geological observations and chemical analysis so that interpretations would include an understanding of the physical setting and processes which have contributed to the weathering fates.

Due to the great diversity of beach environments impacted by the T/V Exxon Valdez oil spill, an abundance of trapped oil pockets exposed to various degrees of energy and biological activity were created. These areas of various exposure and energy are defined as microenvironments. The August 1992 sampling cov-

ered a variety of microenvironments at Knight, Smith, Perry, Latouche, Block, and Crafton Islands. The types of weathered oil found on these beach surfaces ranged from mousse, asphalt pavements, water surface sheens, to rock stains and flake. The subsurface samples consisted of heavy to light oil residue found in boulder to pebble and sand beach material. This range of oil types indicates significantly different physical, chemical and biological degradation processes are occurring and may have been influenced by natural weathering and the beach cleaning techniques. For clear indications and statistical references to the extent of weathering, the samples were analyzed by detailed GC/MS to characterize and source-fingerprint the residual oils collected.

The GC/MS analysis were completed by selective/multiple ion monitoring, focusing on normal alkanes and PAH compounds such as alkylated dibenzothiophenes, phenanthrenes, naphthobenzothiophenes, pyrenes, and chrysenes, fluoranthene, anthracene, and various benzopyrenes. Several of these components are often associated with long-term chronic toxicity. These PAH compounds comprise less than 2% of original T/V Exxon Valdez oil. GC/MS has the sensitivity and capability of selectively analyzing these compounds by the individual peaks or grouping alkylated compounds. The quantitative results from the analysis have been nor-

malized for the compositional differences related to weathering and highlighted for relative toxicities.

All of the samples collected during August 1992 exhibit some degree of weathering. In general thick residual oil without sediments, such as crevice samples, were only slightly weathered as characterized by evaporative loss of the normal alkanes less than nC-12. Close examination of the nC-17/pristane and nC-18/phytane ratios suggest some selective microbial degradation has occurred. Samples of light oil (oily residue) in coarse beach surface material have been highly weathered, as evident by significant alteration of the normal alkanes by microbial degradation. The PAH profile show considerable depletion of the 2-ring naphthalenes and significant reduction of the 3-ring phenanthrenes and dibenzothiophenes; it is only

the C-2 and C-3 alkylated homologues of these compounds that persist. The alkylated naphthobenzothiophene, pyrene and chrysenes appear to be the most persistent aromatic hydrocarbons. Additional contribution of combustion-sourced PAHs were detected in some of the trace level samples.

A general trend is that the rate of degradation is proportional to the concentration of oil in the sample. Subsurface oil and thick oil deposits persist since they are protected from the physical processes which breakup the oil into smaller fragments, creating a greater surface to volume ratio which aids the natural rate of weathering by evaporation, dissolution, photo-oxidation, and biological oxidation (biodegradation). Therefore a major limitation to biodegradation is the availability of oil to the microbial community.



Toxicity of Intertidal and Subtidal Sediments Contaminated by the Exxon Valdez Oil Spill

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This study was conducted under the auspices of the State-Federal Natural Resources Damage Assessment programs. The study was designed to: a) demonstrate and quantify the toxicity of oiled environmental samples, using standard toxicity tests; and b) determine the extent to which any observed toxicity may be attributed to oxygenated, polar products in weathered oil (versus the parent hydrocarbons found in fresh crude).

To estimate the toxicity potential of sediments oiled by the Exxon Valdez oil spill, standardized toxicity tests were applied to intertidal and subtidal sediment samples taken during the cruises of the *Fairweather* in 1989, the *Davidson* in 1990, and *The Big Valley* in 1991. In 1989, the sediment toxicity (EC-50's determined by Microtox[®]) was significantly rank correlated with hydrocarbon concentration, determined by ultraviolet fluorescence (UVF), in intertidal samples from 42 sites in Prince William Sound and impacted portions of the Gulf of Alaska. Toxicity measured by Microtox[®] in subtidal (3-20m) sediments also showed a generally decreasing trend with increasing distance from the spill center, as did hydrocarbon concentrations measured by UVF.

Toxicity was estimated in 1990 (21 sites in PWS and 8 outside) and 1991 (14 sites in PWS) with a sediment elutriate test using larval oysters *Crassostrea gigas*,

and with a whole sediment test using the amphipod *Ampelisca abdita*. The 1990 toxicity tests with amphipods indicated that: 1) intertidal toxicity was substantially greater than subtidal toxicity; 2) mortality was correlated with hydrocarbon concentrations measured by UVF in intertidal sediments, but not at other depths; and 3) mean mortality for intertidal sediments at ten exposed sites inside of Prince William Sound was significantly higher than for six reference sites. Significant amphipod toxicity (relative to controls) was demonstrated in intertidal sediments from the following sampling sites (all notably oiled, listed in order of declining toxicity): Northwest Bay, Snug Harbor, Block Island, Chugach Bay, Chenega Island, Sleepy Bay, and Tonsina Cove. No statistically significant toxicity was detected in any subtidal sediment samples in 1990, and mean mortalities in subtidal sediments were not significantly different between exposed and reference sites.

In 1991, the mortality of test amphipods relative to controls exhibited a lower range than in 1990 (0-50.5%, compared to 0-98.7%), but because control mortality was lower and less variable than in 1990, the threshold for statistically significant differences from controls occurred at lower levels of mortality. Among the eight oiled sites sampled, significant toxicity to amphipods was found at Snug Harbor (6 & 20 m), Sleepy Bay (0 & 6 m),

Northwest Bay (0 & 6 m), Herring Bay (0 & 6 m), Disk Island (0 & 20 m), and Bay of Isles (6 m). Subtidal (6 m) sediments were generally as toxic as intertidal sediments. However, significant toxicity to amphipods was also found at two of the six reference sites sampled: Drier Bay (0, 6 & 20 m) and Mooselips Bay (0 m). Significant mortalities of oyster larvae were also detected with shallow sediments from oiled [Sleepy Bay (0 m), Bay of Isles (0 m), Chenega Island (0 & 6 m) and Block Island (6 & 20 m)] and reference sites [Drier Bay (0 m), Mooselips Bay (6 m), MacLeod Harbor (6 m) and Rocky Bay (6 m)]. As a result, the mean toxicities for the oiled and reference sites in 1991 were not significantly different for either amphipods or oyster larvae. The observed patterns of toxicity are consistent with the general decline of hydrocarbons in the intertidal zone over the period 1989-1991, and with the concomitant transfer of hydrocarbons into shallow subtidal sediments. These results suggest a significant decline in oil-related toxicity between 1990 and 1991, concurrent with the removal and disappearance of the lower molecular weight aromatic compounds.

To explore which fractions of petroleum were potentially most toxic, large samples of intertidal sediments (4 kg) and interstitial porewater samples (114-170 liters) were collected in September 1990 from a heavily oiled site (Bay of Isles on Knight Island) and an unoiled site (Mooselips Bay on Montague Island). The pore waters and sediments were extracted exhaustively with a mixture of methylene chloride and ethyl acetate, and the extracts were subsequently fractionated by liquid column chromatography into aliphatic, aromatic and polar components. Analysis by gas chroma-

tography showed that the water and sediment samples from Bay of Isles contained substantial quantities of petroleum hydrocarbons representing moderately weathered petroleum (as evidenced by absence of n-alkanes below C12 and absence or depletion of mono- and di-aromatic compounds in the aromatic fraction). The pore water sample from Mooselips Bay was essentially free of petroleum hydrocarbons.

The aromatic and polar fractions from both sites were systematically tested with a variety of toxicity tests (Microtox®, SOS Chromotest®, bivalve larval survival and normal development, anaphase aberrations and sister chromatid exchange in developing bivalve larvae, and teratogenicity and anaphase aberrations in salmon embryos) to determine the relative toxicities of the two chemical fractions. Both the aromatic and polar fractions from Bay of Isles sediment samples were consistently more toxic than analogous fractions from Mooselips Bay based on Microtox® and SOS Chromotest®, and abnormality, anaphase aberration, and sister chromatid exchange responses in bivalve larvae. The polar fraction from the Bay of Isles porewater samples also exhibited greater toxicity for most endpoints than the analogous samples from Mooselips Bay. For the aromatic fractions from pore water, however, the differences in test results between the Bay of Isles and Mooselips samples were generally insignificant, and the toxicities approximated that of accompanying method blanks.

In the Bay of Isles samples, the polar and aromatic fractions elicited approximately equivalent toxic responses in the sediment extracts, while the polar fraction was usually slightly more toxic for the porewater extracts. There were no

consistent patterns, however, to distinguish responses to water and sediment samples. For example, EC-50's for the Microtox® response were 10g and 23g (dry weight) of sediment for the aromatic fractions from the two Bay of Isles samples, while those for the polar fractions from the same samples were 8.9g and 65g, respectively. Analogous EC-50's for the two water samples from Bay of Isles were 1150 ml and 445 ml for the aromatic fractions and 154 ml and 393 ml for the polar fractions, respectively. The toxicity test results with the fractionated extracts showed greater differences between Bay of Isles and Mooselips than were found in the 1990 field survey, in which Bay of Isles sediment samples were not significantly more toxic either to amphipods or to oyster larvae than those from Mooselips Bay. This difference probably reflects the heterogeneous distribution of oil in beach sediments. Overall, the results of these toxicity tests on these highly concentrated fractions indicate very low sediment toxicity compared to sediments from industrial or urban areas, where Microtox® EC-50's may be 2-3 orders of magnitude lower.

These toxicity measurements tend to confirm previous observations and conclusions that the acute toxicity in crude oil is caused primarily by low-molecular weight aromatic constituents. In short-term exposures, molar toxicity appears to increase with number of aromatic rings (i.e., benzene < naphthalene < phenanthrene), at least up through 3-ring compounds, and also with the extent of substitution (i.e., benzene < toluene < xylene < ethylbenzene, etc.) (Rice et al., 1977). All of these more toxic constituents are lost during earlier stages of petroleum weathering, and were significantly depleted in Prince William Sound sedi-

ments by the time our samples were collected in fall 1990.

Oxidation products of aromatic compounds in petroleum are produced through microbial metabolism and photooxidation, and intermediary metabolites of polynuclear aromatic compounds are known to be genotoxic. Very little work has been published on these compounds, but some of them could undergo bioaccumulation and exert toxicity to marine organisms. The present studies show, however, that very low genotoxic responses were associated with petroleum in the sediments and pore water from Bay of Isles.

Previous studies also suggested that oxidation products were unlikely to exert significant short-term effects under ambient conditions. For example, although the toxicities of phenol (and p-cresol) were found to be intermediate between those of naphthalene and toluene (naphthalene~5x phenol~2x toluene), Korn et al. (1985) concluded that the phenols were not major contributors to the toxicity of water-soluble fractions (WSF) of oil, because the concentrations of toluene and naphthalene were respectively about 50x and 2x-7x higher than that of phenolic compounds in the WSF. Similarly, Malins et al. (1985) identified oxidized products of phenanthrene, including carbonyl, quinone, and carboxylic acid derivatives, in seawater after UV irradiation of a phenanthrene "slick" for 120 hours.

About half of the oxidation products of phenanthrene in the seawater after this UV irradiation were not extractable with methylene chloride, indicating oxidation to highly water-soluble products. UV irradiation of No. 2 fuel oil in a flow-through, agitated system caused less than a 2-fold increase in total extractable or-

ganic materials (compared to an unirradiated SWAF), and no differences were observed in mortalities of English sole embryos exposed for 48 hours. Irradiation under static conditions, however, enhanced the extractable organic material in the SWAF about 23-fold (to 161 ppm), and substantial mortality occurred, with an apparent EC-50 of about 25 ppm.

Preparation of SWAFs from Prudhoe Bay crude oil under identical conditions, however, produced no differences between flow-through and static conditions either in levels of total extractable organic materials or in mortalities of English sole embryos. Malins et al. (1985) concluded that these studies provided no evidence that photooxidation would under most conditions significantly enhance the toxicity of petroleum in the marine environment.

Along with the results presented here, previous studies suggest that polar con-

stituents, whether present in the parent oil or formed as a result of degradation in the environment, do not pose a significant additional risk of toxicity or mutagenicity to marine organisms.

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Microbial Activity in Sediments following the T/V Exxon Valdez Oil Spill

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Shortly after the grounding of the *Exxon Valdez* on 24 March 1989, the National Oceanic and Atmospheric Administration (NOAA) organized a multi-investigator cruise to document the extent of oil contamination of coastal habitats in Alaska. This first survey cruise was followed by five seasonal cruises over the next 2 years organized as a joint effort of NOAA and the Alaska Department of Environmental Conservation. The purpose of these survey cruises was to document oil concentration distributions and assess the relative ecological impacts of the spill to intertidal and subtidal areas.

Assessment of microbial populations was an important component of the surveys since a major fate of petroleum contaminants in marine environments depends on the ability of microorganisms to use hydrocarbons as a source of carbon and energy (Leahy and Colwell, 1990). Additionally, patterns of hydrocarbon mineralization activity and distribution of hydrocarbon-degrading microorganisms can be used as an indication of *in situ* biodegradation of petroleum (Madsen et al., 1991). Measurements of total numbers of hydrocarbon-degrading microorganisms and assays for the mineralization potential of hydrocarbon fractions by these populations provide evidence of the presence of hydrocarbons that can be utilized by microorganisms. When sediments from a pris-

tine environment are perturbed with oil, this distribution reflects the extent, movement and persistence of the contamination.

We sampled 38 sites within Prince William Sound throughout a three year period following the oil spill. In these samples we measured numbers of hydrocarbon-degrading microorganisms and mineralization potentials of radiolabelled hydrocarbon fractions in shoreline sediments and subtidal surface sediments at depths to 100 m. Depending on the cruise, up to 6 isobaths were sampled for each site; intertidal (0 m), 3 m, 6 m, 20 m, 40 m, and 100 m. At the 0 m, 3 m, 6 m, and 20 m isobaths sediment samples were made up of subsamples collected from eight random locations along a 30 m transect parallel to shore by shore party or SCUBA divers. The 40 m and 100 m samples were collected by Van Veen or Smith-MacIntyre grabs and subsampled from the surface of the sediment. The number of hydrocarbon-degraders in each sample was estimated by using the Sheen Screen most probable number technique which uses disruption of an oil film to indicate the presence of hydrocarbon-metabolizing microorganisms (Brown and Braddock, 1990). Radiorespirometry was used to assay the hydrocarbon-oxidation potential of microorganisms in sediment slurries (Brown et al., 1991). The compounds [$1-^{14}\text{C}$]-

hexadecane, [1,(4,5,8)-¹⁴C]-naphthalene and [9-¹⁴C]-phenanthrene were used as paradigms of aliphatic and polycyclic aromatic hydrocarbons. Hexadecane potentials were determined after 2-day incubations. Two-day naphthalene and phenanthrene potentials were universally very low and reference sites (sites known not to have been oiled by the *Exxon Valdez* oil spill) were generally 0 or near 0 after 10 days for all cruises. For these reasons, 8- or 10-day incubations were used for naphthalene and phenanthrene data. Significant differences (at the 95% confidence level) for numbers of hydrocarbon-degraders or mineralization potentials at a site compared to the reference sites were determined by the Mann-Whitney U Test (Zar, 1984).

The numbers of hydrocarbon-degrading bacteria vary by several orders of magnitude among sites and dates sampled after the *Exxon Valdez* oil spill. Ranges for numbers of hydrocarbon-utilizing bacteria during 1989 in this study were similar to those found for the *Amoco Cadiz* oil spill (Ward et al., 1980). Microbial studies in Alaskan coastal sediments conducted before the *Exxon Valdez* oil spill are limited in number. A 1975-1977 survey of Cook Inlet and the Gulf of Alaska found the highest mean numbers of hydrocarbon-oxidizing bacteria determined by a plate count method to be 8.4×10^3 cells/g dry weight of sediment at a site in upper Cook Inlet near several oil wells (Roubal and Atlas, 1978). These authors hypothesized that sediments containing 10^3 to 10^4 oil-degrading bacteria/g dry weight probably had a previous history of oil exposure from either biogenic or polluting sources.

In the summer of 1989, eleven shoreline sites in Prince William Sound exceeded the maximum value for hydro-

carbon-degraders found in 1978 (Roubal and Atlas, 1978). In fall 1989 all 14 shoreline sites sampled in this study had high numbers of hydrocarbon-degraders, ranging from 3.6×10^3 to 5.5×10^5 cells/g dry weight sediment; reference sites had a median of 38 cells/g dry weight sediment. Statistically significant higher numbers of hydrocarbon-degraders were observed at these oiled sites than at the reference sites. Median numbers of hydrocarbon-degrading microorganisms on the shorelines in Prince William Sound decreased from 1989 through 1991. However, there were still several shorelines in the summer of 1991 that had $> 10^3$ hydrocarbon-oxidizing bacteria/g dry weight sediment. In the summer of 1989, numbers of hydrocarbon-degraders in subtidal surface sediments at depths greater than 6 m were below the detection limits of the assay (< 13 /g dry weight sediment). However, at some sites by the summer of 1990, there were measurable numbers of hydrocarbon-degraders at all depths (beach through 100 m). Data from the summer of 1991 show a trend toward much lower total numbers of hydrocarbon-degraders for all sites and depths, implying that conditions are no longer favorable for biodegradation or that biodegradable hydrocarbons are no longer present.

The median 2-day hexadecane mineralization potentials maintained a fairly constant level through the fall of 1990 and then dropped dramatically by the summer of 1991. In spring and summer of 1990 many sites, even at depth, had potentials for hexadecane mineralization significantly greater than the reference sites. However, in the fall of 1990 only a few sites had potentials significantly greater than the reference sites. By the summer of 1991 potentials of hexadecane

mineralization were low at all sites. The reduction of the hexadecane mineralization potentials may be due to a decrease in numbers of microorganisms acclimated to hydrocarbon biodegradation or a decrease in the hexadecane remaining in the sediment, or some combination of the two factors.

Median potentials of polycyclic aromatic hydrocarbons (PAH) oxidation increased with time from the summer of 1989 reaching a maximum in 1990 and then dropping to much lower levels in 1991. The potentials for phenanthrene mineralization were slightly greater when mineralization potentials for naphthalene and phenanthrene were run on the same sediment samples (Fall, 1989). This finding is supported by a previous study of polluted sediments in Boston Harbor, Massachusetts, where naphthalene turnover times in the Harbor were found to exceed those for phenanthrene (Shiari, 1989). The difference in potentials between phenanthrene and naphthalene seen in Prince William Sound is unlikely to exclusively account for the increase in mineralization potentials of PAH between summer of 1989 and summer of 1990. Mineralization potentials for phenanthrene remained high through the fall of 1990 but declined substantially by the summer of 1991. The data for the summer of 1991 show that there were still many sites with high phenanthrene oxidation potentials relative to the reference sites. However, the absolute values for mineralization potential were much lower than for previous cruises.

The objective of our study was to document the impact of the Exxon Valdez oil spill on the population and activity of hydrocarbon-degrading microorganisms in sediments in Prince William Sound.

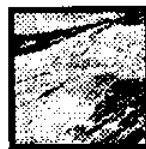
The numbers and activity of these microorganisms are good indicators of exposure of sediments in Prince William Sound to hydrocarbons and may be useful indicators of the mobilization of hydrocarbons with time. The increase of numbers of hydrocarbon-degraders compared to likely pre-spill values, coupled with high mineralization potentials for hexadecane and phenanthrene, also provide evidence of rapid acclimation of naturally occurring microbial populations for biodegradation of these compounds in most sediments.

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Contamination of Subtidal Sediments by Oil From the *Exxon Valdez* in Prince William Sound, Alaska

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The purpose of this project was to assess the degree of petroleum hydrocarbon contamination of subtidal sediments from 32 locations in Prince William Sound resulting from the *Exxon Valdez* oil spill. In this paper we summarize some geographical, bathymetric and temporal trends resulting from analysis of data collected during the first 2 years following the oil spill.

We sampled sediments intertidally and at five subtidal depths in the range 0-100 m in summer and 0-20 m in spring and fall. Shallow sediments (0-20 m) were collected by beach teams or divers on 30 m transects laid along the appropriate isobath. Deep sediments (40 and 100 m) were collected with grabs. Triplicate composite sediment samples were collected at each station. Sediments were sampled in May, July, September and November 1989 and in June, July and September 1990. Oil concentrations, reported in mg/g (ppt), are estimates of equivalent concentrations in sediments of original (fresh) *Exxon Valdez* oil and are based on a weathering model developed in conjunction with principal components analysis (PCA; Short and Heintz, these proceedings). *Exxon Valdez* oil concentrations less than 25 µg/g sediment are not detected using this model because corresponding concentrations of individual polynuclear aromatic hydrocarbons on which the model is based approach analytical method detection limits of about 1 ng/g.

Geographical distribution:

Petroleum hydrocarbons were found to have contaminated subtidal sediments over a broad geographic range in Prince William Sound from the north end of Eleanor Island to southern Elrington Island. Lower intertidal (0 m) sediments were contaminated by *Exxon Valdez* oil at no fewer than nine locations in 1989 and 12 locations in 1990. Subtidal sediments were contaminated by *Exxon Valdez* oil at no fewer than 12 locations where oil had come ashore (oiled locations) in 1989 and 1990. Those locations constituted 67% of oiled locations studied in 1989 and 86% of those oiled locations studied in 1990. Contamination of subtidal sediments by *Exxon Valdez* oil at oiled locations reached a depth of at least 20 m at seven sites in 1989 and at 14 sites in 1990.

Bathymetric distribution:

The greatest concentrations of *Exxon Valdez* oil in benthic sediments were found in the lower intertidal region (0 m). An average concentration ($n=3$) as high as 24.7 mg/g was found at 0 m on Disk Island in July 1989. The greatest concentrations of *Exxon Valdez* oil in subtidal sediments were found at the shallow depths. The highest concentration recorded was 5.2 mg/g in a sediment sample collected at about 3 m at Sleepy Bay in September 1989. Concentrations of *Exxon Valdez* oil exceeding 1.0 mg/g in subtidal sediments occurred at nine locations in 1989 and reached a depth of 20 m at Eshamy Bay in July 1989. How-

ever, no significant difference was found in the concentration of oil between depths at oiled sites in July (when all depths were sampled). In 1990 the highest concentration of oil in subtidal sediments recorded was 4.3 mg/g at 20 m at Fox Farm in September. Concentrations of oil exceeding 1.0 mg/g were found at two locations (Herring Bay and Fox Farm) reaching a depth of 20 m at both sites.

In July 1990 when all depths were again sampled oil concentrations in sediments at oiled sites were greater ($p < 0.01$) at 3 and 6 m than at greater depths. The majority of sediments from 40 and 100 m were found not to be contaminated with *Exxon Valdez* oil. Where contamination was found it was at relatively low concentrations (≤ 0.43 mg/g, 1989; ≤ 0.45 mg/g, 1990) of oil.

Temporal distribution:

Examination of temporal changes in the contamination of sediments by oil revealed that detectable quantities of *Exxon Valdez* oil moved over time to shallow subtidal depths at locations with heavily oiled shorelines. At Sleepy Bay no significant trend was seen in the concentration of oil in sediments at mean lower low water (0 m) between May 1989 and September 1990. Nevertheless, over the same period of time subtidal sediments at 3, 6 and 20 m in Sleepy Bay showed increasing oil concentrations to a peak concentration followed by a decline to levels comparable to early post-spill levels or less. At the 3 m depth the peak ($p < 0.05$) occurred in September 1989. The peak occurred in November 1989 at 6 m ($p < 0.005$) and 20 m ($p < 0.01$) and persisted at 6 m until June 1990.

At Northwest Bay and Herring Bay also there was some evidence that oil

moved to greater depths over time. Northwest Bay and Herring Bay showed significant decreases ($p < 0.01$ and $p < 0.001$ respectively) in the concentration of oil in sediments at mean lower low water (0 m) between May 1989 and September 1990. At both sites, the oil concentrations in sediments collected at 3 m did not change significantly between May 1989 and September 1990. At Northwest Bay the concentration of oil in sediments at 6 m peaked ($= 0.95$ mg/g, $p < 0.05$) in September 1989, whereas oil concentrations in sediments at 20 m showed no significant change during the study period. At Herring Bay no significant change occurred in oil concentrations in sediments at 6 m during the study period, but concentrations changed at 20 m reaching a maximum ($= 0.55$ mg/g, $p < 0.05$) in September 1990.

Oil was detected in subtidal sediments at a number of locations in Prince William Sound but concentrations were markedly less than in sediment samples from heavily oiled intertidal sites. Oil became broadly distributed in subtidal sediments during the first 2 years following the *Exxon Valdez* oil spill. Oil concentrations attained their highest values in the low intertidal and shallow subtidal (0-20 m) regions.

Sediments collected at 40 and 100 m were for the most part not contaminated with *Exxon Valdez* oil. There was some indication that some movement of oil down slope took place at heavily contaminated sites. Although oil concentrations in subtidal sediments were probably not acutely toxic to most organisms, the low-level oil concentrations were widespread, persistent over the 2-year period, and would be a source of chronic exposure to subtidal communities.

Determination of Petroleum-Derived Hydrocarbons in Seawater Following the Exxon Valdez Oil Spill I: Analysis of Seawater Extracts

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We analyzed samples of subsurface seawater within Prince William Sound following the Exxon Valdez oil spill, to evaluate the extent of water contamination by petroleum hydrocarbons. These samples were collected in three groups beginning 1, 3, and 5 weeks following the oil spill. Triplicate samples were collected from depths of 1 and 5 meters at the 30 locations sampled, which ranged from heavily oiled locations to control locations that were not affected by the spill. All three of the triplicate samples were analyzed from the first group of samples collected, but only one of the three triplicate samples was analyzed from each of the subsequent two groups. Each 900 ml seawater sample was extracted twice with a total volume of 75 ml dichloromethane within 5 minutes of initial collection, then stored at -20°C until analysis.

These samples were analyzed using single ion mode gas chromatography-mass spectrometry (GCMS/SIM) for the most abundant 2 to 5 ring polynuclear aromatic hydrocarbons (PAH's) in the spilled oil, and using gas chromatography-flame ionization detection for alkane hydrocarbons including pristane, phytane, and the normal alkanes of 10 to 30 carbon atoms (C_{10} to C_{30}).

During the first sampling period, summed PAH's were highest at sampling stations adjacent to beaches that were heavily contaminated by the spilled oil. Summed PAH concentrations ranged up to $6.60 \pm 0.62 \mu\text{g/L}$ seawater (95%

confidence interval, $n = 3$) at Snug Harbor, and ranged from $1.92 \pm 0.40 \mu\text{g/L}$ to $5.23 \pm 1.27 \mu\text{g/L}$ at sampling stations near heavily oiled beaches of Northwest Bay, Herring Bay, southeast Eleanor island, north Smith island, and the Bay of Isles. These summed PAH concentrations include PAH's from any source, and do not distinguish PAH's associated with particulate oil and dissolved PAH's. Elevated PAH concentrations were also detected at several more open-water sites between Knight and Montague islands. At all these sites, summed PAH concentrations were slightly higher at the 1 m depths than at the 5 m depths.

In contrast, summed PAH's were lowest at sampling stations that were near the margin or else were distant from the path of the spilled oil through the Sound. PAH concentrations typically ranged from 0.4 ± 0.2 to $1.5 \pm 0.6 \mu\text{g/L}$ seawater at these locations.

The relative concentrations of individual PAH's differed markedly among the sampling sites. At sites near the margin or distant from the path of the spilled oil, naphthalene was the predominant PAH compound detected, accounting for 40% to 100% of the summed PAH's found. Although naphthalene was consistently detected at both 1 m and 5 m at these sites, other PAH's were only sporadically detected at concentrations near detection limits.

However, at sites near heavily oiled beaches, or at the more open-water sites between Knight and Montague islands

where elevated PAH concentrations were found, numerous PAH's were simultaneously detected at concentrations substantially above detection limits. Naphthalene accounted for generally less than 40% of the PAH's found at these sites, and the proportion decreased with increasing summed PAH concentrations.

At sites near heavily oiled beaches, or at the more open-water sites between Knight and Montague islands where elevated PAH concentrations were found, relative concentrations of detected PAH's are very highly correlated with corresponding relative PAH concentrations of *Exxon Valdez* crude oil. Product-moment correlation coefficients of PAH's in *Exxon Valdez* crude oil and in samples from these sites generally range from 0.85 to 0.95 ($P < 0.001$) with 14 to 18 PAH's included in the correlation (but with naphthalene excluded). Also, PAH's that are absent or present at low relative concentrations in *Exxon Valdez* crude oil were not detected in these samples.

We conclude from these results (1) that *Exxon Valdez* crude oil is the proximate source of PAH's in samples where measured PAH's are elevated and strongly correlated with *Exxon Valdez* crude oil PAH's, and (2) that an additional source of naphthalene is present in all samples, suggesting an unknown sampling contamination source for naphthalene only. The first conclusion derives from the close association of samples containing elevated PAH concentrations with areas directly impacted by the spilled oil, the strong correlation of relative PAH concentrations in these samples and in the spilled oil, the general absence of these PAH's in samples from areas marginal or distant from the path of the spilled oil, and the absence of a plausible alternative explanation of these observed

results. The second conclusion derives from the ubiquity of naphthalene at a minimum apparent concentration of about $0.4 \mu\text{g/L}$ seawater; a similar pattern of naphthalene detection persisted in the second and third sampling periods.

We estimated total PAH's attributable to *Exxon Valdez* crude oil in samples where PAH's are strongly correlated. This estimate is the sum of measured PAH's except naphthalene; plus an amount of naphthalene proportional with measured amounts of 1- and 2-methylnaphthalene, consistent with this same proportion in *Exxon Valdez* crude oil. In every case, the naphthalene calculated in this manner as attributable to *Exxon Valdez* crude oil is less than the measured amount of naphthalene in the sample.

Exxon Valdez oil PAH's (EVO-PAH) are quantitatively parallel with summed PAH's: EVO-PAH concentrations ranged up to $6.24 \pm 0.63 \mu\text{g/L}$ seawater at Snug Harbor, and ranged from $1.26 \pm 0.40 \mu\text{g/L}$ to $4.72 \pm 1.18 \mu\text{g/L}$ at sampling stations near heavily oiled beaches of Northwest Bay, Herring Bay, southeast Eleanor island, north Smith island, and the Bay of Isles. Elevated EVO-PAH concentrations were also detected at several more open-water sites between Knight and Montague Islands. At all these sites, EVO-PAH concentrations were generally somewhat higher at the 1 m depths than at the 5 m depths.

EVO-PAH concentrations generally declined with time. At most sites, EVO-PAH concentrations declined by a factor of 2 or more from the first to the second sampling period, and by more than a factor of 2 from the second to the third sampling period. Exceptions included sites where oiled beach clean-up activities had commenced, such as at Herring

Bay or Snug Harbor, where EVO-PAH concentrations increased slightly by the second sampling period at some depths. EVO-PAH's were not evident at any open-water site after the first sampling period, and were generally less than 1 $\mu\text{g/L}$ seawater at any site by the third sampling period.

Comparison of EVO-PAH concentrations and C_{19} through C_{27} n-alkane concentrations suggests the presence of at least some particulate oil in the samples that contained EVO-PAH's. To estimate relative proportions of dissolved and particulate EVO-PAH's, we calculated an aromatic hydrocarbon enrichment factor (AHEF) as the ratio of EVO-PAH measured in a sample, and the minimum expected EVO-PAH based on measured C_{19} through C_{27} n-alkane hydrocarbons. At sites where PAH's are strongly correlated with Exxon Valdez PAH's, this AHEF ranged from 0.69 to 5.99. Values of this AHEF near 1 are consistent with particulate oil, whereas values substantially above 1 indicate dissolved PAH. However, this AHEF index is not rigorous indicator of particulate oil, because values near 1 may by chance be due to dissolved EVO-PAH's present with odd carbon-numbered alkanes derived from natural sources.

The relative concentrations of dissolved EVO-PAH's suggests they are determined by dissolution kinetics, and not by solubility of individual PAH's. Relative dissolved PAH concentrations that correlate strongly with those of Exxon Valdez crude oil, suggests that the composition of dissolved PAH's matches that of the oil. The initial relative rates of dissolution of individual PAH's are determined by differences among individual PAH's of molecular attractive

forces to the seawater phase and to the remaining oil phase, and by the relative concentrations of the PAH's in the oil. Differences of these attractive forces among PAH's are approximately proportional with molecular surface area, which varies by less than a factor of 2 among the EVO-PAH's (naphthalene through C-4 phenanthrene), whereas the relative concentrations of these compounds in the oil vary more than 30-fold. Initial dissolution rates are therefore mainly determined by relative concentrations in the oil. In contrast, solubility is not an important factor because concentrations attained by dissolved PAH's are well below solubility limits, and the volume of affected seawater in Prince William Sound is much greater than the volume of the spilled oil, thereby guaranteeing that solubility limits of EVO-PAH's are never approached.

Although readily detectable, these concentrations of EVO-PAH's are well below levels that are acutely toxic to marine fauna. On the other hand, these data demonstrate that PAH's from Exxon Valdez crude oil were available to subsurface marine fauna the first few weeks following the oil spill, especially in near-shore, near-surface waters that are particularly productive areas biologically. In addition, if mononuclear aromatic hydrocarbons had been measured in addition to the PAH's that were measured, total aromatic hydrocarbon concentrations in the seawater column derived from spilled Exxon Valdez oil would almost certainly have been higher, possibly exceeding the State of Alaska water quality standard of 10 $\mu\text{g/L}$ seawater, because mononuclear aromatic hydrocarbons are much more abundant than PAH's in crude oil.



Methods for Determining Crude Oil Contamination in Sediments and Biota After the Exxon Valdez Oil Spill

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The grounding of the *Exxon Valdez* on March 24, 1989 spilled almost 11 million gallons of Prudhoe Bay crude oil (PBCO) into the waters of Prince William Sound, Alaska. As part of the Natural Resource Damage Assessment effort, thousands of samples of sediment and biota were collected to determine the distribution of the spilled crude oil and the exposure of the marine animals. Therefore, the use of rapid, low-cost analytical methods, generally known as screening methods, to estimate concentrations of petroleum-related aromatic compounds (ACs) in samples was vitally important to the production of timely information in the emergency response. Screening methods can rapidly process large numbers of samples to provide a semiquantitative estimate of contaminant concentrations and thus, allow ranking of samples by degree of contamination. Accordingly, the most contaminated samples can be located by screening; then, detailed analyses, e.g., gas chromatography/mass spectrometry (GC/MS), can be focused on the selected samples to confirm the presence of contaminants. Screening for metabolites of ACs in fish and marine mammals

Thousands of samples of fish and marine mammals were collected from the *Exxon Valdez* spill area to determine the exposure of these animals to PBCO. Because fish and marine mammals extensively metabolize most ACs in their livers and the metabolites are transferred

to bile for excretion, AC metabolites must be measured in these animals to establish their exposure to PBCO. Concentrations of metabolites were estimated in bile of fish and marine mammals using a reverse-phase high-performance liquid chromatographic (HPLC) screening method that measured fluorescence at wavelength pairs specific for 2- and 3-ring petroleum-related ACs (Krahn et al. 1992). Then, GC/MS was used to validate the HPLC screening results by measuring concentrations of individual metabolites of petroleum-related ACs, e.g., alkylated naphthols and phenanthrols, in these animals. Because the concentrations of metabolites measured by HPLC screening and sums of AC metabolites from GC/MS were highly correlated, the bile screening method was validated as an effective tool for estimating concentrations of AC metabolites. Screening for crude oil in sediments

Following the *Exxon Valdez* spill, thousands of sediment samples were collected to determine the degree and distribution of the oiling. Because analyzing all these samples by GC/MS would be excessively expensive and time-consuming, priorities for analyses needed to be set. Therefore, a size-exclusion HPLC method used previously to measure AC contaminants in urban sediments (Krahn et al. 1991) was employed to determine concentrations of PBCO in more than 400 sediment samples. Sediments from a large number of sites in the spill area

were surveyed and many were found to be contaminated by PBCO (Krahn et al. submitted). Similar to the results for bile, summed concentrations of individual ACs in the sediments determined by GC/MS were found to be highly correlated with the concentrations of ACs measured by HPLC screening method. Thus, the utility of the rapid HPLC screening method has been extended to analyzing sediment samples for the ACs characteristic of crude oil, thereby directing priorities for GC/MS analyses. As a result, the overall costs of the analyses have been reduced, while still providing the necessary detailed data in a timely fashion.

Establishing source of contamination by HPLC and GC/MS analyses

When fish or marine mammals were exposed to PBCO in the field or injected with PBCO in the laboratory, the chromatographic patterns were similar, but some differences were also apparent. Variations in bile chromatographic patterns can occur because of variations in the degree of exposure of individual animals to the oil or to species-specific differences in metabolism of the petroleum ACs.

In addition, physical factors from the chromatographic process itself, such as the chromatography column used, the condition of the column or the acidity of the mobile phase, can affect the appearance of a chromatogram in reverse-phase HPLC. Therefore, due to both the variability of the metabolic process in various fish species and to the variability of the reverse-phase chromatography of the metabolites, the HPLC chromatographic pattern of bile can only suggest the type of contamination. However, the source of contamination can often be

established by examining GC/MS results. For example, evidence of PBCO contamination in fish and marine mammals was provided by identifying high proportions of certain bile metabolites (i.e., the alkylated naphthols, phenanthrols and dibenzothiophenols) that result from the metabolic conversion of ACs that are characteristic of PBCO (Krahn et al. 1992).

The HPLC chromatograms from sediment are easier to interpret than those from bile. These chromatographic patterns are less variable than those from bile, because the size-exclusion chromatography is stable and because sediment screening measures the ACs themselves. Many of the confounding factors present in the bile chromatograms due to species-specific differences in degree of metabolism or excretion of metabolites are not found in the sediment chromatograms. However, HPLC chromatographic patterns were not consistent among all the extracted sediments from the Prince William Sound area. The differences were not due solely to the degree of weathering of the crude oil, but reflected different sources of ACs, e.g., crude oil or diesel fuel (Krahn et al. submitted).

For example, the chromatograms of Herring Bay and Knight Island sediments, two sites that were heavily oiled, were nearly superimposable with those from weathered PBCO. Furthermore, the chromatographic patterns from the Herring Bay and Knight Island sediments were very different from those of other sources of contamination (e.g., diesel fuel or marine lubrication oil) that might be found in Alaskan sediments. In contrast, results from screening sediments from MacLeod Harbor and Olsen Bay, sites not in the direct path of the

spill, revealed low concentrations of ACs and an HPLC chromatographic pattern that resembled that of diesel fuel. The contaminant source suggested by HPLC screening of these sediments could often be confirmed by comparing the identities and proportions of the ACs determined by GC/MS to similar characteristics of the probable sources. For example, evidence for PBCO as the source of contamination in many Prince William Sound sediments (e.g., those from Herring Bay and Knight Island) was provided by identifying in these samples the high proportions of the alkylated naphthalenes, phenanthrenes and dibenzothiophenes that are characteristic of this crude oil (Krahn et al. submitted).

The HPLC screening methods have important roles in evaluating anthropogenic contamination in samples of bile and sediment. First, samples containing AC contaminants can be rapidly ranked by degree of contamination and second, HPLC chromatographic patterns can provide a basis for suggesting possible

contaminant sources. As a result, expensive GC/MS resources can be effectively allocated. This approach—combining HPLC screening for petroleum-related ACs or their metabolites in sediment and bile with confirmation of contaminant concentrations in selected samples by GC/MS—has proven useful in establishing the extent of damage to natural resources following the *Exxon Valdez* oil spill.

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Qualitative and Quantitative Determination of Exxon Valdez Crude Oil in Sediment Samples Using Principal Component Analysis of Hydrocarbon Data

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We have developed a model for sediment hydrocarbon data that may be used to (1) distinguish *Exxon Valdez* oil contamination from other sources of hydrocarbons, (2) estimate the original amount of *Exxon Valdez* oil in sediments when present, and (3) determine the relative degree of weathering of a sample. The model is derived from an assumption of first-order loss kinetics of each of the hydrocarbon analytes employed, where principal component analysis is used to identify a weathering pattern that is characteristic of spilled *Exxon Valdez* crude oil. When applied to consistent sediment hydrocarbon data sets derived from samples that were collected from known oiled beaches, one principal component

accounts for more than 96% of the data variance. First-order rate constants for hydrocarbon analyte losses were estimated using this data subset, and the relative magnitudes of these constants indicates that the weathering process is predominantly kinetically controlled, where the rate of loss of aromatic hydrocarbon analytes decreases with extent of alkyl substitution.

Use of this model as an interpretive aid and as a unifying framework will be presented, together with results of the application of the model to hydrocarbon data derived from over 2,200 Natural Resource Damage Assessment sediment hydrocarbon samples collected from 1989 through 1991.



Nearshore Subtidal Transport of Hydrocarbons and Sediments Following the Exxon Valdez Oil Spill

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From 1989-1992, subtidal sediment traps were placed in Prince William Sound to capture settling organic and mineral particulate matter offshore of oiled and unoled shorelines. As a component of water quality, settling particulates were collected to: (1) determine if petroleum hydrocarbons were present; and (2) to learn more about subtidal sediment transport processes affecting sediments at the study sites.

Sedimentation of hydrocarbons is a rapid and important fate of spilled oil. Estimates of accumulation in the subtidal include 8-10% of the *Amoco Cadiz* oil off the Brittany Coast (Gundlach et al., 1983) and 10-15% of the unrecovered *Tsesis* oil in the Swedish archipelago (Johansson et al., 1980). Oil can sink by adsorption to sediment, possibly by electrostatic bonding to fine-grained clay micelles (Bassin and Ichiye, 1977), and through uptake by zooplankton and subsequent deposition in fecal pellets (Conover, 1971). Salinity, clay mineralogy, and the presence of organic matter (which may mask adsorption sites) can affect the adsorption (Meyers and Quinn, 1973).

Oiled beaches act as a reservoir from which hydrocarbons may be removed by erosion and deposited in offshore sediments. Ten percent of the oil stranded on untreated shorelines after the Baffin Island Oil Spill project was transported into shallow offshore subtidal sediments

(3-7 m depth) (Boehm et al., 1987). While the bulk of stranded shoreline oil was removed from Prince William Sound beaches in 1989 and 1990, by a combination of treatment activities, natural physical processes and biodegradation, a number of locations have intertidal subsurface oil lenses that are still fluid (at least during summer months) and that are persistent because of protection from surface weathering processes.

Persistence and mobilization of spilled oil is related to physical and biological processes, such as wave, tide and wind energy, microbial degradation and bioturbation. Sediment grain size and oil quantity and composition are among many interacting variables (Blount, 1978; Gundlach et al., 1978). Oil eroded from contaminated shorelines and entering the water column may settle in the nearshore subtidal or move into deeper waters before settling depending on particle size and shape, settling velocity, wave energy, tidal current velocities, and longshore currents (Gundlach et al., 1978). After settling on the benthic surface, oiled sediments can continue to be moved by bottom currents, resuspended by wave-induced oscillatory currents, or be buried deeper into the benthic sediments by bioturbation.

A laboratory study by Bragg et al. (1990) using oiled sediments from Prince William Sound shorelines found that the formation of an emulsion of micron-sized

mineral particles, polar components of oil residue and seawater impeded the adhesion of oil to the larger rocks on the shorelines, allowing natural removal by waves and tidal flushing. Bragg concluded that since the emulsion floc particles were composed mainly of seawater and fine-grained sediments, they would be transported great distances before settling and would be widely dispersed. While this is supported by Stokes Law of settling velocities for individual fine sediment grains, sediments may be trapped in estuaries as physicochemical flocculation with other particles in the water column produce settling rates an order of magnitude greater than the individual grains (Drake, 1976; Kranck, 1975).

Methods

Sediment traps have been used after oil spills to monitor settling particulates for oil contamination (*Tsesis* spill in 1977, Johansson et al., 1980) and to determine sedimentation rates in embayments (Lund-Hansen, 1991) and the open ocean (Woods Hole, 1989). For this study, base-mounted sediment traps consisting of PVC pipe, 15 cm in diameter and 1.2 meters tall were deployed at sites in Prince William Sound representing a variety of oiling and coastal conditions. The traps were placed at 10, 15, and 20 meter water depths offshore of oiled and unoiled shorelines. Divers retrieved and redeployed the traps at approximately 3 month intervals from November 1989 through mid-March 1992. Sediments were immediately filtered from the traps on the vessel and samples were frozen for later hydrocarbon chemistry and grain size analysis.

Benthic core samples were taken at each of the sediment trap locations to evaluate the sedimentary processes at work at the sites, such as erosional and

depositional events, and define background hydrocarbon concentrations and depth of petroleum hydrocarbon contamination. Sediment samples from the upper 2 cm of benthic sediments around each trap were taken to evaluate grain size distributions and hydrocarbon chemistry. The relative contribution of subtidal transport processes (bed-load, saltation and suspension) (Visser 1969; Middleton, 1976) and delineation of erosional and depositional events (Sundborg, 1956) will be estimated by evaluation of grain size distributions and inspection of the sediment cores.

In addition, approximations of wave energy at particular sites are being calculated from hindcasts using the Automated Coastal Engineering System (U.S. Army Corps of Engineers, 1991) and wind data from the National Oceanic and Atmospheric Administration and National Weather Service stations in Prince William Sound. Results from the hindcasts will provide wave parameters from which bottom stresses can be derived (Komar, 1974). Bottom stress calculations combined with grain size data will allow estimations of the likelihood of sediment resuspension and transportation by waves.

Results

Two years after the spill, elevated concentrations of petroleum hydrocarbons were consistently found in trapped suspended particles near initially heavily oiled shorelines. Sediments retrieved from the traps at five sites in August 1990 showed petroleum hydrocarbon patterns consistent with *Exxon Valdez* crude oil, with the highest concentrations at heavily oiled Sleepy Bay, and lowest at the unoiled control site in Port Fidalgo, indicating an association of petroleum hydrocarbons and trapped sediments with

oiled shorelines.

This association persists in sediments captured over the winter of 1990-91 at 13 sites (retrieved in March 1991). The pattern of hydrocarbons is substantially altered however, with consistent and substantial enrichment of chrysenes relative to the other aromatic hydrocarbon classes at each trap location. While the reasons for this alteration of aromatic hydrocarbon patterns is not well understood, the pattern could reflect complex weathering processes.

The highest concentrations of petroleum hydrocarbons found in March 1991 were in trapped sediments from offshore of heavily oiled locations in Northwest Bay, Sleepy Bay and Snug Harbor, while the lowest concentrations were at unoiled or lightly oiled locations in Eshamy Bay, Stockdale Harbor and Port Fidalgo. This pattern again demonstrates a clear association between concentrations of petroleum hydrocarbons in trapped sediments and degree of oil impact on the adjacent shoreline. Several sites with high concentrations in trapped sediments have subsurface oiling: a persistent lower-intertidal subsurface lens of fluid oil (documented by ADEC shoreline surveys as late as June 1992) remains at Northwest Bay, and significant subsurface oil has been noted on shoreline segments in Sleepy Bay. Benthic sediment samples collected adjacent to the sediment traps in August 1990 also indicate elevated petroleum hydrocarbon concentrations at trap sites adjacent to oiled shorelines. Further conclusions await the results of remaining hydrocarbon analysis.

Hydrocarbon chemistry analysis of the remaining sediment trap, benthic and core samples will be completed by January 1993. Results of grain size, minerology and organic carbon analyses

of the sediment trap and benthic samples are currently being evaluated and will be related to the hydrocarbon results for each site and deployment period to correlate sedimentology with petroleum hydrocarbons. Benthic sediment core stratigraphy will be evaluated for background chemistry and any depositional events. Grain size frequency distributions are being evaluated for an understanding of transport mechanisms at each site.

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Short-term Biological Effects of Shoreline Treatment on Intertidal Biota Exposed to the Exxon Valdez Oil Spill

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A substantial amount of the 11.6 million gallons of Alaska North Slope crude oil spilled from the *T/V Exxon Valdez* on March 24, 1989, was deposited on beaches in Prince William Sound. Following the spill, biological studies were conducted on the biota of the intertidal and shallow subtidal habitats in the Sound to determine short-term effects of several shoreline treatment techniques considered for beach cleanup. Four treatment methodologies examined were high-pressure hot-water (HP-HW) and low-pressure warm-water (LP-WW) wash, and applications of a dispersant (Corexit 7664) and a beach cleaner (Corexit 9580 M2). These methods were designed to remobilize oil that coated the substrate and facilitate its removal from the beaches. Only LP-WW and HP-HW treatments were employed on a routine basis in the Sound.

The basic objective of these studies was to assess and compare the short-term biological impacts of several alternative treatment methodologies. The major elements of these studies, conducted on three islands at the north end of the Knight Island Archipelago in protected boulder/cobble habitat, were pre- and post-treatment measurements of abundance, cover, and community composition of the biota at specific levels in the test areas. These studies employed a stratified-random design with replicate quadrat sampling before and after implementation of the specific shoreline treat-

ment methodologies. Each program compared two treatment alternatives; the programs were independent and not compared at the time.

In all test series, each test plot was exposed to a different treatment. Contrasted tests were sequential rather than simultaneous. Test plots extended from upper to lower intertidal but treatment was supposed to be restricted to mid and upper levels, where most oil came to rest during the initial stranding. We surveyed at least at mid and lower elevations in each test plot.

The purpose of this paper is to present data from several treatment effect studies and provide a general summary of short-term biological effects of treatment. Secondly, we are presenting a qualitative assessment of the validity of the studies themselves.

The relevant questions to be addressed by these studies include: (1) Was treatment accompanied by biological damage? (2) Was the damage caused by thermal, chemical, or physical effects? (3) Which type of treatment caused the least amount of damage? and (4) Was the observed damage long-lived?

An operative assumption for these studies was that the variables examined would not change significantly over the 3 to 10 days between the pre- and post-treatment surveys except as a consequence of the treatment regime associated with the studies. We applied this

assumption to comparisons of all variables examined, ranging from oil cover and cover intensity to community descriptors (e.g., number of species and abundance of individuals) and species abundance.

Our approach was to compare pre-treatment data for variables such as algal and epifaunal cover, epifaunal density, average number of taxa per quadrat, species diversity, and density of dead epifauna, and species composition with post-treatment measurements from the same transects. Additionally, in response to thermal impacts, we measured cover by dead *Fucus gardneri* at Herring Bay following the HP-HW treatment. We applied a stratified-random design stratified on the basis of elevation. We designated mid and lower intertidal strata at all sites based on the dominant biological assemblages; an upper intertidal stratum was included in the Corexit 7664 study at Ingot Island.

Types of effects observed included reductions in density and obvious mortality due to abrasion, crushing, or thermal exposure. Generally, all treatments in the upper and mid intertidal appeared to reduce cover by live algae but HP-HW treatment at Herring Bay and LP-WW treatment in the Corexit 9580 M2 test appeared to cause the most damage ($p < 0.01$). The greatest decrease in algal cover occurred in the middle intertidal, where treatment and high algal cover overlapped. Changes in algal cover in the lower intertidal zone were not significant.

Epifaunal cover in the upper and middle intertidal, highest in the middle intertidal zone, was commonly 50 percent lower following treatment, but generally low cover and high small-scale variability confounded the statistical sig-

nificance of the changes. Epifaunal cover in the algae-dominated lower intertidal was quite low and stable.

Epifaunal density was generally higher at lower intertidal levels. Strong decreases (100-fold) occurred in the HP-HW treatment tests in the middle intertidal ($p < 0.01$). A significant decline following the 165-minute HP-HW treatment at the lower level at Herring Bay reflects reduced density of a hermit crab and a periwinkle. Density changed inconsistently following the other types of treatment, suggesting that impacts from those methods were not strong. Moderate but insignificant increases in density in both tests at Ingot Island seem to represent an increase in abundance of scavengers like hermit crabs, probably in response to the increased availability of dead or damaged organisms.

The average number of taxa per quadrat tended to decline following treatment. Strongest declines occurred at both tide levels at Herring Bay following HP-HW treatment ($p < 0.01$).

We measured density of dead epifauna to provide an indication of mortality in animals such as crabs and mussels. Density was highest and changes were stronger at mid intertidal levels where mussels were most abundant. Large increases in dead animals observed following HP-HW treatments reflect higher numbers of gaping attached mussel shells with intact tissues, suggesting a strong impact from HP-HW treatment. Density of dead animals decreased following treatment in all other tests, probably reflecting the tendency of the wash activities to flush loose materials from the areas.

Several taxa displayed substantial changes in abundance during the tests. Significant changes for algae were mostly

declines that probably resulting from vigorous washing. In a few cases, algal cover increased significantly, possibly because of increased visibility following washing.

Many of the significant changes observed in invertebrates probably reflect removal or relocation caused by the vigorous washing. Barnacles, periwinkles, limpets, mussels, hermit crabs, and a whelk declined significantly at some sites. At other sites, density of periwinkles, a whelk, and a hermit crab increased significantly, probably as a reflection of relocation by the treatment or immigration in response to increased abundance of dislocated prey.

Abundance of dead specimens increased substantially for two species at one site. Density of dead mussels and cover by dead *Fucus* increased significantly at mid levels at Herring Bay following both HP-HW treatments. HP-HW treatment caused *Fucus* to change from a normal olive-green color to blackish orange, the color of dead *Fucus* in drift wrack accumulations. HP-HW treatment for 165 minutes caused twice the mortality as 95-minute exposure.

The type and number of significant changes varied considerably by elevation and type of treatment, probably reflecting the position of the zone relative to washing activities and rigor of washing. Far more of the significant changes were decreases. Significant changes were far more common at the mid and lower tide levels than at upper level but 80 percent of the significant changes at upper levels were declines. Live organisms were generally less abundant following treatment at the mid levels although periwinkles increased considerably at some locations. In contrast, changes at lower levels were more evenly distributed be-

tween decreases and increases.

The relative frequency of significant changes in abundance varied considerably by type of treatment. Corexit and LP-WW treatments were accompanied by relatively few decreases whereas nearly all changes observed following HP-HW treatment were decreases.

Several flaws in sampling design weakened the ability of the studies to evaluate treatment effects. Comparability of substrate and biota in paired test plots was weak. For example, algal cover at the Disk and Ingot Island sites and epifaunal cover, beach slope, and sediment composition at Disk Island varied considerably between the Corexit and LP-WW test plots.

While much of the protected shoreline in Prince William Sound is mixed-soft substrate, systematic evaluations of treatment effects were not conducted in these habitats. In the one instance where the study area included soft substrate, density of hardshelled clams declined from about 160/sq. m. to about 40/sq. m. overnight. This 75-percent reduction in density was probably a consequence of physical effects but because similar habitat was not present in the paired test plot, the cause of mortality is unclear.

Overall, the data suggest that the effects of both chemicals were less severe than those caused by LP-WW or HP-HW treatment. The data, our observations, and a review of the study design also indicate that LP-WW wash accompanying tests of chemical efficiency was less rigorous than when performed by itself. The tests were not comparable in terms of temperature regimes, duration, or consistency of coverage.

Because of the proximity of the paired sites and timing differences in testing, the probability of cross-test interactions

is high. Such interactions could explain several declines in variables at Disk Island where two pre-treatment surveys were conducted.

The absence of reference sites to evaluate the "No-treatment" alternative constitutes a major design flaw since no data exist to support the operative assumption for the tests, i.e., changes observed during the various tests did not occur universally throughout the Sound at unoiled sites.

In summary, the various treatments were accompanied by biological damage, including reductions in density and obvious mortality due to abrasion, crushing, and thermal exposure. While the type and degree of damage varied by type of treatment, some damage accompanied all types of treatment examined. Damage was manifest as (1) a significant degree of reduction in one or more community or population attributes; or (2) increases in the percentage of dead mussels or *Fucus*. Severity and persistence of effects varied by type of treatment. However, the programs were not designed in a manner allowing discrimination among potential causes of damage.

The greatest damage appeared to be a response to thermal exposure or physical/mechanical disturbance. HP-HW wash, resulting in high mortality in algae and epifauna, caused the most severe

and persistent effects. Effects of LP-WW washes were less severe and persistent and cursory data on temperature gradients downslope and across the test plots suggest that LP-WW treatment probably did not cause significant thermal impacts in the intertidal biota. The dispersant and beach-cleaner applications caused few apparent short-term effects. However, flaws in sampling design constrain the degree to which we can extrapolate from the conclusions.

Observations of displacement and mortality in clams at Disk Island and mussels at Ingot Island suggest that physical effects are substantial. Treatment dislodged or excavated many specimens that subsequently became crushed or moribund. Treatment with the dispersant and beach cleaner was accompanied by fewer significant changes in species abundance or community attributes than other methods. However, based on other studies (Houghton et al., 1991), the "no-treatment" alternative appears to produce fewer short- and long-term changes and faster recovery.

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Growth and Survival of the Predatory Snail *Nucella lamellosa* in Areas Exposed to the Exxon Valdez Oil Spill

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Populations of the predatory snail *Nucella lamellosa* (frilled dogwinkle or drill) were studied in Prince William Sound to determine growth and survival at Oiled, Oiled and Cleaned, and Un-oiled sites. In 1991, individual tags were used to mark animals in resident stocks where possible, or, at sites where populations had been decimated, to tag animals imported from a reference population (Hogg Bay). Tagging was done during April/May and July 1991.

Samples of animals tagged in May were measured in July 1991 and samples from most sites were again measured in September 1991 and July 1992. Size changes over periods of up to 14 months were used to evaluate growth differences among treatments and recapture rates were used to estimate survival. This study provides insights regarding the efficacy of treatment following an oil spill and complications associated with possible restoration efforts for species with direct development following a major environmental perturbation.

The Brody-Bertalanffy model with seasonal adjustment was used to describe growth (Sager 1982) and recast as a difference equation using size pairs (S_t and $S_{t+\Delta t}$), a time interval (Δt) and Julian day/365 (t):

$$S_{t+\Delta t} = S_{\infty} - (S_{\infty} - S_t)e^{-K(\Delta t)} \left[1 + \frac{(1-\epsilon)K}{2\pi} \sin 2\pi(t + \Delta t - t_A) \right] \quad \text{Eq.1}$$

with parameters:

S = asymptotic size

K = the growth rate constant

ϵ = parameter measuring strength of the seasonal effect; equal to 1 with no effect.

t_A = parameter that adjusts the time of minimum growth.

Numbers of size pairs (S_t and $S_{t+\Delta t}$) at each location are: 1. Un-oiled sites: Bass Harbor (pairs = 295), Crab Bay (401), Eshamy Bay (360), Hogg Bay (420) and Outside Bay (484); 2. Oiled sites: Crafton Island (239 from Hogg), Herring Bay (32 local and 47 from Hogg), and Snug Harbor (72 from Hogg); and, 3. Oiled and Cleaned sites: Block Island (266 from Hogg), Northwest Bay (418 local and 255 from Hogg), and Smith Island (47 from Hogg).

Parameters in Eq. 1 were estimated by nonlinear regression. The parameters ϵ and t_A both were close to 0, indicating a strong seasonal effect with minimum growth close to January 1; therefore differences in growth among treatments were focused on just K and S_{∞} . The parameters K and S_{∞} are highly correlated and so comparisons among treatments were made using their product ($K \times S_{\infty}$), termed ω (Gallucci and Quinn 1979, Appeldoorn 1983), which has dimensions of cm/yr and approximates the instantaneous growth rate of a newly hatched individual; the larger the value of ω , the faster the growth.

For Unoiled sites (N=5), w ranged from 38.04 - 52.08 with a mean of 44.09. For Oiled sites (N=3) w ranged from 16.60 - 32.56 with a mean of 24.04 and at Oiled and Cleaned beaches (N=3), w ranged from 21.08 to 48.93 with a mean of 36.85. Differences among treatments was tested by ANOVA (N=11) with a resulting $p=0.049$. There was no overlap of w values for Unoiled and Oiled sites. Estimates of w for snails at Oiled and Cleaned sites were in between and overlapped the w values of *Nucella lamellosa* at Unoiled sites and Oiled sites.

Survival of *Nucella* was estimated from the recapture of tagged animals at all of the same sites as growth was determined with the exception of Smith Island where animals were not sampled in July 1992. Two time periods were used: April/May 1991 to June/July 1992 and July 1991 to June/July 1992. At some sites, tagged animals were released both in April/May 1991 and in July 1991. Survival (S) from tagging in either April/May or July 1991 to July 1992 was adjusted to an annual rate (S_A) by

$$S_A = S^{\frac{1}{\Delta t}} \quad \text{Eq.2}$$

where Δt is the difference in years from tagging to July 1992, which ranged from 0.953 to 1.175.

For Unoiled sites (N=5), annual survival probability ranged from 0.123 (Hogg Bay) to 0.319 (Eshamy Bay) with a mean of 0.217. For Oiled sites (N=3), annual survival rate ranged from 0.044 (Snug Harbor) to 0.060 (Herring Bay) with a mean of 0.054. At Oiled and Cleaned sites, the two estimates were 0.108 (Block Island) and 0.114 (Northwest Bay) with a mean of 0.111. Differences among treatments were tested by ANOVA (N=10) following an arcsin transformation of survival probabilities

with a resulting $p=0.005$. There was no overlap of annual survival rates for any of the treatments.

Estimates of annual survival must be taken as minimum values because there probably were some tagged animals living at each site that were not found in July 1992. Additional sampling would be necessary to obtain better estimates of the actual numbers present at the final census; however, there is no reason to suspect bias in the samples with respect to treatment. Given the errors associated with variable effort in collections at the sites, the trend of survival rates probably is correct: best survival at Unoiled locations, worst at Oiled sites that were not cleaned, and intermediate at Oiled and Cleaned sites.

Trends of survival are similar to trends shown in growth in the sense that *Nucella lamellosa* at Unoiled sites generally had both better growth and survival than animals at Oiled or Oiled and Cleaned sites. Growth and survival at Oiled and Cleaned sites, in general, appeared to be better than at Oiled sites that were not cleaned. However, the numbers of sites is small and clearly number of study sites for each treatment should be increased in future studies of these and other populations recovering from oil damage.

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Recovery of Prince William Sound Intertidal Infauna from Exxon Valdez spill and Treatments-1990-1992

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Much of the crude oil spilled from the tanker *Exxon Valdez* on March 24, 1989, was deposited on beaches in Prince William Sound. Major beach cleanup activities began in May and continued throughout the summer of 1989. About 400 km of shoreline were treated in the sound in 1989 using various hydraulic wash and bioremediation (fertilization) techniques; additional mechanical cleanup and bioremediation occurred during the summers of 1990 and 1991.

High pressure, hot-water (HPHW) washes used on mixed gravel/sand/silt beaches in 1989 altered the nature of habitat available to infauna. Hydraulic washing of heavily oiled upper beach areas transported large quantities of silts, sands (to 4 mm diameter), and even pebbles (4 to 64 mm) down the face of the beach to the water's edge. Coarser materials were deposited on the lower beach, while suspended oils and silts were carried from the area by currents in both surface and water column plumes. Presumably many organisms, along with a large proportion of the organic matter in the sediment column, were similarly displaced.

The overall objectives of this study were to evaluate recovery of important intertidal habitats and resources from the effects of oiling and shoreline treatment and to assess the influence of HPHW treatments on the nature and rates of

recovery (Houghton et al. 1991, 1992). The study plan established was designed, in part, to document persistence of effects of 1989 hot-water washes, if they remained evident, over the broader area where hot-water treatments had been applied. Primary variables isolated in the sampling design were habitat type, tidal elevation, degree of oiling, and use of high pressure, hot-water shoreline treatments. This paper reports sampling of mixed gravel/sand/silt (mixed-soft) beaches that had been unoiled (reference or Category 1 sites), oiled but not treated with HPHW washes (Category 2), and oiled with subsequent HPHW-wash treatment (Category 3). Information on initial oiling and on shoreline treatments applied at our study sites were derived from State of Alaska and Exxon records and through contacts with on-site personnel.

Quantitative field surveys were conducted in Prince William Sound in mid-summer (late June to early July) 1990, 1991, and 1992 to document environmental conditions and infaunal assemblages at 9 to 12 intertidal sites in mixed-soft habitats. Two stations were established at each site to represent intertidal elevations (zones) of biological interest. At each station (elevation), five sediment cores (0.009 m² by 15 cm deep) were randomly collected. Samples were field sieved on 1.0-mm screens and preserved.

In the laboratory, all infauna were identified to the lowest practicable taxon. Additional samples were taken at each station for sediment hydrocarbon analysis (all years), grain size determination (1991 on), and total organic carbon (TOC) and total Kjeldahl nitrogen (TKN) analysis (1992 only). Samples of littleneck clams (*Protothaca staminea*) were collected for age and growth analysis and for analyses of tissue hydrocarbon content. Littleneck clams also were tagged and transplanted between sites with differing residual sediment hydrocarbon levels in May 1991. Clams were recovered in September 1991, and survival, growth, and bioaccumulation of hydrocarbons were evaluated as a function of sediment hydrocarbon concentration.

Study results confirmed that protected sand and gravel beaches were severely affected by hydraulic treatments that greatly altered beach morphology. Coarse sands and fine gravels were flushed from upper intertidal elevations and often buried the lower beach in several centimeters of sediment. In this process, many infaunal organisms along with a high percentage of the silts and organic materials in the sediments were dislodged and transported from the site. Hydraulic treatments left the lower beaches in many areas covered with coarse sediments with a low content of fines. TOC and TKN were lowest in sediments at HPHW-washed lower stations.

Distribution patterns of polycyclic aromatic hydrocarbons (PAHs) in mixed-soft sediments in 1990 and 1991 were similar: PAHs were significantly (to three orders of magnitude) lower at unoiled (Category 1) sites than at oiled (Category 2), or oiled and HPHW-washed (Cat-

egory 3) sites. PAH concentrations at Category 2 sites were lower at the lower elevations and highest at upper elevations in 1990. By 1991 substantial weathering had occurred at middle and upper elevations, and concentrations were reduced by an order of magnitude; little weathering had occurred at lower stations. In contrast, at Category 3 sites PAH concentrations were lower at upper and lower intertidal elevations and higher at mid- and subtidal elevations. By 1991 only very slight reductions in PAH concentration had occurred at Category 3 intertidal stations, but subtidal concentrations had dropped by an order of magnitude.

These patterns suggest that although shoreline HPHW treatment has resulted in an initial drop in oiling in intertidal sediments (in 1990 Category 2 sites had greater average intertidal PAH than Category 3), it has not made a dramatic difference in overall hydrocarbon concentrations. Ten-fold declines were observed at three of four elevations in Category 2 but at only one of four elevations in Category 3. Based on the degree of replication included in the averages, these changes, indicative of considerable weathering, are fairly reliable. The most prevalent constituents in 1991, in order of importance, were compounds of dibenzothiophene, phenanthrene, and naphthalene. In contrast, the most prevalent compounds in 1990 were naphthalenes, phenanthrenes, fluorenes, and dibenzothiophenes. Reduced concentrations of naphthalenes and fluorenes are another reflection of the weathering process.

Several compounds occur at sufficiently high concentrations in sediments at some stations to raise concerns about

sublethal effects of exposure. Processes that could be influenced by chronic exposures to low levels of PAH include survival, reproduction, development, and growth. In the clam transplanting experiment (carried out in 1991), there was a clear correlation between higher sediment hydrocarbon concentrations and reduced survival of littleneck clams.

In 1990, 1991, and 1992, infauna at lower mixed-soft stations appeared only moderately affected by the spill on Category 2 (oiled but untreated) beaches; significant differences between Category 1 (unoiled) and Category 2 stations were few. The infauna on Category 3 (oiled and hot-water-washed) beaches, however, was fundamentally altered in comparison to both other beach categories. Number of species, number of organisms, and species diversity varied significantly among station categories; lowest values were at the HPHW-treated beaches in all 3 years. Most major taxa (gastropods, bivalves, polychaetes) had significantly lower abundances on Category 3 beaches than on Category 1 and/or 2 beaches in 1990 and 1991. In 1992 these relative abundances remained unchanged but were no longer significant indicating that some recovery is under way.

In 1991 and 1992 several dominant taxa were most abundant at the lower intertidal station at the heavily oiled Category 2 site at Block Island. This area continued to show extremely high sediment oiling yet had higher densities of the deposit-feeding bivalve *Macoma* spp., harpacticoid copepods, and oligochaetes than any site group. These taxa may be capable of exploiting hydrocarbon-degrading bacteria in these oily sediments.

The Block Island lower station also had a high density and the highest re-

cruitment of young-of-the-year clams despite the fact that sediment hydrocarbon concentrations were sufficient to cause significantly reduced survival and increased tissue PAH uptake in clams experimentally transplanted to this station. Interestingly, the survival of littleneck clams in the 1991 transplant experiment was highest (98%), and growth rate was greatest, at the Northwest Bay West Arm site (Category 3), which has had very clean sediments since 1990; this site has had consistently low clam recruitment compared to Block Island, however. In all three years, the Category 3 sites had the lowest overall density and lowest recruitment rates of hardshelled clams (both littlenecks and butter clams, *Saxidomus giganteus*).

Analysis of infauna data confirms that the effects of shoreline treatments relate as much to physical disturbance (burial, displacement, reductions in fines and organic content) as to oiling. Infaunal assemblage variables (total organism density, diversity, richness) were negatively correlated with percentage of sands and the residual hydrocarbon levels in the sediments in 1991; total organism density was positively correlated with the percentage of fines.

The 1990-1992 data indicate that recovery of infauna on hot-water-washed beaches will take many years. Primary factors prolonging the recovery period on Category 3 beaches are the continued instability of the beach profile, reduced content of finer sediments (including organics), reduced recruitment, and destruction of the normal population (age) structure in longer lived organisms such as the hardshelled clams. Residual sediment oiling may also alter the pathway to, and delay the realization of, full recovery at least one Category 2 lower

station (Block Island). Multivariate analyses confirmed patterns of category differences and trends in recovery.

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Recovery of Prince William Sound Intertidal Epibiota From the Exxon Valdez Spill and Treatments—1990-1992

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Following the 1989 Exxon Valdez oil spill, significant concerns were raised regarding the potential effects on intertidal habitats and biota of high pressure, hot-water (HPHW) washes used to remove oil from the shorelines of Prince William Sound. The objectives of this study were to evaluate recovery of important intertidal and shallow subtidal habitats and resources from the effects of oiling and shoreline treatment and to assess the influence of HPHW treatments on the nature and rates of recovery. This study also was designed to extrapolate persistence of effects of 1989 hot-water washes over the broader area where hot-water treatments had been applied. Primary variables isolated in the sampling design were habitat type, tidal elevation, degree of oiling, and use of HPHW shoreline treatments. The status of recovery of intertidal assemblages from the oil spill and subsequent shoreline treatments was examined by repeated sampling of a suite of rocky intertidal sites during 1990, 1991, and 1992.

Studies sponsored by Exxon in 1989 demonstrated that major intertidal assemblage dominants (rockweed, mussels, limpets, snails) survived 3 to 4 months in heavily oiled habitats. Immediately following HPHW washing, however, these taxa suffered significant reductions (50 to 100 percent losses; $p < 0.1$; Houghton et al. 1991). Because of these identified adverse impacts, ecological effects of this type of treatment were a

major focus for the present research effort.

We sampled multiple rocky shores that had been unoiled (Category 1 sites), oiled but not treated with HPHW washes (Category 2), and oiled with subsequent HPHW-wash treatment (Category 3). Information on initial oiling and on shoreline treatments applied at our study sites was derived from State of Alaska and Exxon records and through contacts with on-site personnel.

Stratified random sampling was used to assess epibiota (surface dwelling plants and animals) at nine to 15 intertidal rocky sites (depending on the survey) representing several exposures and degrees of disturbance in selected oiled and unoiled locations in the sound. Two to three stations were established at each site to represent intertidal elevations (zones) of biological interest. At each station, multiple 0.25-m² quadrants were randomly located, permanently marked, and sampled to document the abundance of (surface dwelling plants) and fauna (animals). Samples of selected organisms were collected for analyses of age and growth and tissue hydrocarbon concentrations.

This sample design allowed for monitoring long-term recovery trends at sites of known oiling and treatment history. It is also well suited (by the level of replicated sampling at each station) for comparisons, at specific points in time, between pairs of stations with similar habi-

tat but different oiling and/or treatment histories. Because of the limited number of stations that could be sampled in each habitat/oiling/treatment category, this design is less well suited to statistical inference regarding the generalized impacts of oiling and treatment over all stations with similar histories. Nevertheless, statistically significant differences among site categories were shown for some variables in all sampling years. These results have allowed us to draw conclusions regarding initial impacts and directions of recovery.

Our studies in 1990 provided strong evidence of bioaccumulation at several levels in the food web but found no evidence of biomagnification. PAH concentrations decreased from lower to higher levels of the food web and were lowest in the top predators examined. Moreover, relationships between PAH concentrations in prey and potential predators from the same site were weak. On the basis of these findings, collections of the sunstar (*Pycnopodia helianthoides*) and the drill (*Nucella lamellosa*) for PAH analysis were discontinued in 1991.

PAH analyses in 1991 focused on determining whether high concentrations of PAH in mollusk tissues at some sites were due to continued exposure to hydrocarbons or to residual hydrocarbons in the tissues from exposure during previous years. These analyses produced three important findings:

1. PAH concentration in tissues of mussels (*Mytilus cf. trossulus*) transplanted from reference sites to areas of high residual sediment contamination increased over the summer by an order of magnitude or more to levels of contamination as high as, or higher than, those in resident (local) animals. Levels of tissue PAHs in mussels

(transplants and local animals) at Smith Island (3.7 to 20.4 ppm dry), considered one of the more highly contaminated sites remaining in the sound, were similar to the levels of PAHs in mussels from near Seward (6.2 ppm dry), a reference site.

2. Levels of contamination observed in resident mussel tissues at Smith Island in July and September 1991 had dropped more than an order of magnitude from those observed at that site in July 1990. The composition of PAHs in mussel tissues was quite similar to that seen in 1990 but reflected weathering in the source hydrocarbons. Phenanthrenes and dibenzothiophenes were dominant; naphthobenzothiophenes and fluorenes were of intermediate importance; and naphthalenes, pyrenes, and chrysenes were of low importance.
3. The most likely sources of long-term contamination of mussel tissue in the Sound are the reservoirs of subsurface oil at many sites. Large reductions in PAHs in mussel tissues from Smith Island suggest that leaching rates from such subsurface deposits of oil have declined dramatically since July 1990, however. This observation is important in consideration of the advisability of continued shoreline treatment activities, particularly in view of the fact that, by 1991, the tissue contamination at Smith Island had declined to a level similar to that observed in animals from near Seward.

Distribution and Abundance of Epibiota

In our 1990 and 1991 sampling, a high degree of variability was seen among

biota at sites subjected to varying degrees of treatment. Many of the important longer-lived dominants remained intact at some treated sites in 1990; in other areas, apparently those that had been cleaned more rigorously, these species did not survive. In 1991, recolonization of these areas was evident on most rocky shorelines. Trends in the initial impact and recovery of three key taxa (rockweed, limpets and drills) at all middle elevation rocky stations sampled illustrate that oiled but untreated (Category 2) stations were well on their way to recovery by mid-1991—that is, there were no longer significant differences in abundance between Category 1 and Category 2 stations.

Biota at HPHW-washed (Category 3) middle rocky stations, however, remained significantly depressed; mean abundances of rockweed, limpets, and drills (*Nucella* spp.) were very low at Category 3 rocky sites through May 1991. Mean abundance of these taxa all showed partial recovery at HPHW sites by July 1991.

By July 1992, rockweed cover and densities of limpets at HPHW middle stations exceeded those at unoiled stations. Full recovery had not yet occurred, however. Littorine snails were least abundant at Category 3 stations in 1990; in July 1991 the density of the opportunistic *Littorina scutulata* increased sharply at Category 3 sites and in 1992 was much more abundant at the HPHW stations. Its congener *L. sitkana*, which lacks planktonic larval dispersal, was slower to recolonize hot-water-washed stations. Drills were virtually absent at all heavily oiled middle stations, both treated and untreated, through 1992 but remained at relatively constant low densities at lightly oiled or unoiled sites

over the same period.

Lower rocky intertidal areas in Prince William Sound tend to be dominated by longer lived algae because predation by seastars and drills greatly limits the numbers of grazers. Only one HPHW-treated lower rocky station has been tracked for all 3 years of the study. In 1990 and the spring of 1991 this station had a high percent cover of ephemeral green and brown algal colonizers and limited populations of grazers. By July of 1992 this site had higher densities of limpets and littorine snails than the other lower rocky stations (HPHW), perhaps because of reduced numbers of seastars and drills. A primary apparent effect of HPHW washing was the reduction of longer lived red algae, especially Delesseriaceae, Gigartinaceae, Palmariaceae, and Rhodomelaceae. These groups accounted for 15.3 percent cover in 1990 and declined to 8.8 percent by 1992 at the single HPHW lower rocky site sampled in all three years. At Category 1 and 2 lower stations, this group of algae ranged between 35.1 to 48.3 percent cover over the same period.

One rocky site in Northwest Bay that was stripped bare by treatments in 1989 showed little colonization at middle and upper stations through September 1991. Films of blue-green algae and possibly other algae that developed early in 1990 and 1991 were grazed or eroded away; mostly bare rock was left. Even early successional colonization by rockweed sporelings, *Fucus gardneri*, or the barnacle *Semibalanus balanoides*, such as that seen elsewhere over broad areas, occurred only sporadically and in isolated patches. By 1992, these patches had expanded to cover portions of the middle station where crevices in the rock retain moisture; the remainder of the station,

which lies on a smooth rock bench, continued to be devoid of significant algal or animal growth. Clearly, the epibiota at this site will take many more years to recover to pre-spill conditions.

Dense stands of young rockweed that had been evident only as inconspicuous sporeling mats in some HPHW areas in 1990 were growing well in 1991 and gave a superficial appearance of a "normal" shoreline. A more detailed examination, however, revealed that the assemblage in these heavily treated areas bore little semblance to that on Category 1 (unoiled) or 2 (oiled but not hot-water-washed) shores. Longer lived, more stable components of the upper rockweed zone (rockweed; several red algae, Rhodophyta; hermit crabs, *Pagurus hirsutiusculus*; a limpet, *Lottia pelta*; and drills, *Nucella lamellosa*) were significantly less abundant in the HPHW-treated areas. Rockweed on many HPHW-washed sites in 1991 was composed predominantly of 2-year-old plants that were not reproductively mature and a small percentage of sporelings. In contrast, oiled sites that were not HPHW-washed included an even mix of several year classes of older and reproductively mature

plants. By 1992, rockweed established as sporelings on Category 3 sites in late 1989 was reproductively mature.

Thus, oiled rocky areas not subjected to severe cleanup activity were generally indistinguishable from unoiled sites in 1990 and 1991; more heavily treated sites remained in earlier stages of recovery through 1991. By 1992, most HPHW sites were also well on their way to recovery; abundances of most taxa were similar to those on unoiled rocky beaches. Multivariate analyses confirmed patterns of category differences and trends in recovery. Qualitative examinations of other shorelines around the northern portions of the Knight Island group in July 1991 revealed many areas where early successional assemblages were present in areas where hot-water treatments were used in 1989. The broader ecological implications of resetting the successional stage over large areas of shoreline are not clear.

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NOAA's Long-Term Ecological Recovery Monitoring Program: Overview and Implications of Recovery Trends and Treatment Effects

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NOAA's Hazardous Materials Response and Assessments Division is responsible for providing, through the Scientific Support Coordinator, guidance to the Federal On-Scene Coordinator during responses to major oil spills. The aim of that guidance is to maximize protection of marine resources. A major source of guidance comes from experiences gained during previous spill responses. Obviously, the *Exxon Valdez* oil spill provided many opportunities for learning about the success and failures of various containment, removal and shoreline treatment methods. This paper summarizes the rationale and approach of NOAA Hazardous Materials Response and Assessments Division shoreline treatment studies in Prince William Sound and suggests some implications for future responses and restoration.

Focus on Shoreline Treatment

One of the most controversial and poorly quantified aspects of oil spill response is the extent to which shorelines should be treated to remove oil. By May, 1989, much of the spilled crude oil was stranded on 357 miles of protected and exposed shoreline in Prince William Sound where it became an actual or perceived long-term threat to commercial and subsistence fisheries, wildlife and recreation/tourism. Acknowledging cautions expressed by the Scientific Support Coordinator and other groups, the decision was made to use any and all means practical to clean the shorelines, including high energy methods such as high-pres-

sure hot-water washing. After considerable debate, agencies also agreed to leave a few shorelines untreated to use as reference sites provided studies be conducted to monitor impacts and recovery.

During the summer of 1989 shoreline clean-up involved unprecedented use of high-pressure (50-100 psi) hot- or warm-water (to 140F) washing. Warm-water washing became the primary method for nearly all treated shoreline in Prince William Sound (Exxon Production Research Corp., 1990). Initial tests on several types of shoreline indicated that this method removed most of the shoreline surface biota that otherwise survived oiling (Lees and Houghton, 1990 and Houghton et al, 1991). Thus a central question of the HMRAD program was the extent to which high-pressure hot water washing enhanced or delayed oil removal, return of altered shoreline integrity and especial recovery of shoreline marine communities.

Hazardous Materials Response and Assessments Division Programs

During 1989 and 1990 NOAA's Hazardous Materials Response and Assessments Division initiated chemical, geomorphological and biological monitoring programs at over 35 sites in Prince William Sound to help document the benefits and effects of various shoreline countermeasures. Primary objectives of this multifaceted effort were (1) to determine the initial effects of oiling and treatment on the nearshore environment; (2) to determine whether or not treatment enhanced

mine whether or not treatment enhanced or delayed recovery; (3) to characterize the nature of physical and biological recovery on a long-term basis; and ultimately, (4) to provide response and restoration agencies with guidance for appropriate and effective actions.

Components of the NOAA/HMRAD program include (1) biological monitoring, (2) shoreline geomorphology, and (3) oil weathering studies. The core biological program samples shorelines representing three treatment categories: unoiled and untreated; oiled but not high-pressure hot-water washed; and oiled and high-pressure hot-water washed. Oiling and treatment histories at each site were extracted from State and federal treatment records, derived from personal contacts and directly observed from historical videos and photographs.

Three shoreline types were sampled: (1) rocky, (2) cobble/gravel/mud, and (3) boulder/cobble. There is a minimum of three sites for each combination of treatment and shoreline type, and each site generally includes visual, biological and chemical surveys at each of three elevations (upper, middle and lower intertidal). Five to ten replicate measurements are made to document changes in surface oiling, epibiota and infauna at each elevation at least annually. Keystone species of mollusks are tagged, released and recovered to document improvements or impacts on growth. Composite samples of sediments, mussels and clams are sampled to document trends in concentrations of hydrocarbons. The geomorphology program (Michel et al, 1991) determines the wave exposure regime at each site and documents changes in beach profiles and trends in the abundance and character of oil both at the surface and below. In 1992, a

special chemical study was undertaken to evaluate differential weathering in various microhabitats on Prince William Sound beaches and the implications for bioavailability.

The program was not part of the Natural Resource Damage Assessment (NRDA) process, but rather, an extension of response activities. Nevertheless, it provides information and insights highly relevant to any assessment of physical or biological recovery in Prince William Sound. Moreover, the program has a number of unique features that distinguish it from other research efforts in Prince William Sound. Treatment is explicitly accommodated as an analytical variable, incorporating the use of untreated shoreline segments ("set-aside" sites) as reference areas. The program also integrates observations of biology, bioaccumulation, chemical fate, shoreline geology, and coastal processes to provide a physical framework for observed biological conditions and trends. The NOAA program characterizes trends in shoreline ecosystem components not otherwise addressed in NRDA programs, e.g. intertidal infauna, and growth, recruitment and hydrocarbon contamination of clams. Finally, the program is conducted under a regimen of rigorous, open, multi-agency peer review.

Implications for Response and Restoration

Results of biological and some chemical studies through 1992 are presented in several reports at this conference (Houghton et al, 1993a and 1993b) and elsewhere (Michel and Hayes, 1992; Houghton et al, 1992). Below, we suggest some of the implications of results to date.

Did high-pressure hot-water washing enhance or impede shoreline recov-

ery? The answer depends on how we define recovery and what kind of shoreline is involved. It clearly reduced the amount of visible oil on all shoreline types. Below the surface, however, it is difficult to distinguish washed from untreated shorelines. Although the amount of subsurface oil has decreased each year, subsurface oil in various stages of weathering remained at all oiled and treated shorelines in the summer of 1992 (Michel and Hayes, 1992).

Furthermore, unoiled sites were not necessarily free of petroleum hydrocarbons - all reference sites had evidence of PAH's from combustion or other sources, so there is no pristine background to return to. It also appears there are and will be some long-lasting effects of washing on the structure of some shorelines where contaminated sediment was washed from the upper to lower intertidal and even out into open water. The abundance and diversity of shoreline marine life and the abundance of specific major plants and animals was significantly reduced by high pressure hot water washing, compared to unoiled sites and sites not receiving this treatment (Houghton et al, 1992). As a result, the return of these populations to reference values has been faster at the untreated sites than at the treated sites. Sampling in 1992 showed that although recolonization and recovery are progressing at most sites, the process is far from complete (Houghton et al, 1993a and b).

How long will recovery take?

While it is very difficult and also very risky to predict the long-term course of ecological recovery based on limited temporal information, the data from 1990 - 1992 suggest that at treated sites recovery - i.e., return to abundances at unoiled sites - of various intertidal populations

may take from three to over 15 years. Projections have not yet been made for complete loss of subsurface oiling or return to normal sediment structure.

Implications for Restoration

Since sub-surface oil remained at many sites in the summer of 1992 there may be a natural inclination for additional treatment. The 1992 Technical Advisory Group concluded that additional treatment would be counter productive. Michel and Hayes (in prep) conclude that subsurface oil concentrations are declining at about equal rates at treated (washed and berm relocation) sites. Mussels in some areas continued to be contaminated, but it is not clear that underlying sediments, the presumed source of contamination, are any more contaminated than sediments not underlying mussel beds. Thus it is not clear, without experimental evidence, if additional intervention will enhance reduction of contaminant loading.

We made several interesting observations that may have implications for enhancing restoration of intertidal communities. High-pressure hot water washing almost totally eliminated dogwhelks (*Nucella* spp.), major intertidal predators. Since they do not have planktonic larvae and cannot recolonize from planktonic settlement, they are extremely slow to return. However, transplanted stocks have survived well and grown rapidly at washed shorelines, suggesting transplanting is a feasible restoration activity. Likewise, we have observed high survival and growth of clams transplanted into areas where they were severely reduced or eliminated.

Recommendations for Future Responses

High energy treatments, such as high-pressure hot-water washing may have a

role in the arsenal of shoreline countermeasures. However, if the clean-up strategy includes minimizing damage to shoreline resources, this method should only be used on a case-by-case basis.

Implications for monitoring

The work done to date underscores the need to continue the monitoring effort into the future in order to properly characterize the complex processes that comprise ecological recovery. Monitoring directed at evaluating treatment alternatives is an important part of responses to future spills. However, it cannot be done properly without establishing untreated set aside areas. The concepts of accurate treatment record-keeping and set asides should be considered essential elements in any major spill response.

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The Effects of the Exxon Valdez Oil Spill on Benthic Invertebrates in Silled Fjords in Prince William Sound

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During October 1989, when examining the effects of the Exxon Valdez oil spill on several shallow subtidal habitats in western Prince William Sound, high mortality of invertebrates and fishes was observed in a heavily oiled silled fjord. Here we present the results from the initial observations and sampling of this fjord in 1989, and subsequently in 1990 and 1991. Other silled fjords were examined in 1990. We consider the effects of the spill versus natural benthic hypoxia or anoxia.

The main silled fjord studied is a small embayment in northeastern Herring Bay which is located along the northwestern side of Knight Island. It has an area of approximately 600 km² with the greatest depth within the basin approximately 35 m; the sill depth at the entrance of the fjord is only 4 m. At depths greater than 10 m the substrate was mainly composed of fine, flocculant silt. This fjord was considered as "heavily oiled" after two shoreline surveys were conducted during the summer and fall of 1989.

The dead and moribund animals observed in 1989 were primarily in the deeper portions (> 13 m) of the fjord. In one area extensively surveyed (approximately 70 m²), we observed over 40 dead animals laying on the bottom, including 23 large polychaete worms and 11 starfish (all *Pycnopodia helianthodes*) and miscellaneous clams. Also encountered were dead mobile organisms, such as shrimp, squid and Pacific cod. In addition to the

observed dead organisms, the substrate had a patchy, cobweb-like layer of the bacteria, *Beggiatoa*. This colorless, sulfur-dependent, hemolithotrophic bacteria is associated with decaying vegetation and low dissolved oxygen.

Immediately following these observations, we collected samples of infaunal invertebrates from three randomly placed transects along each of three depth strata (0-2, 2-8, 8-20 m) using a diver-operated suction dredge. One randomly placed 0.25 m² quadrat was sampled to a substrate depth of 10 cm at each depth stratum. A video of each transect was made by divers and a bathymetric chart of the embayment was made using a fathometer aboard a small boat. Additional videos were taken along transects through the deeper portions of the fjord in order to document the extent of dead organisms.

Diver observations and density estimates of infauna were again made from this fjord in May and October 1990 and August 1991. In addition, two more silled fjords (one oiled and one unoled) were examined in May 1990; four (two oiled and two unoled) were examined in September 1990. All sites had features similar to Herring Bay. We first conducted a bathymetric survey at each site as described above. Three stations were then established at random positions along the 20 m-depth contour at each site. At each station, divers collected duplicate 0.1 m² suction dredge samples of sediment for benthic infauna. Measurements

of temperature, salinity and dissolved oxygen were taken on all surveys, except October 1989.

Although the benthic community in Herring Bay in 1989 was obviously stressed, as noted by the dead animals, it still contained a relatively rich assemblage of infauna (e.g., 24 taxa at the family and higher taxonomic level). However, signs of disturbance were evident in the moderately low Shannon diversity index (1.7), the moderately high Simpson dominance index (0.4), and the near absence of sensitive burrowing amphipods (16/m²). The dominance index was mainly attributed to stress-resistant taxa such as the bivalves *Lucina tenuisculpta* (Lucinidae) and *Mysella tumida* (Montacutidae), and the polychaetes *Nephtys cornuta* (Nephtyidae) and *Polydora socialis* (Spionidae).

Lucina and other lucines appear to be able to live where conditions are extreme and oxygen and food limited (Yonge and Thompson, 1976). *Lucina* is in the same order as the stress-tolerant *Thyasira* genus, and several species of *Thyasira* have been reported from organically enriched and polluted substrates (Pearson and Rosenberg, 1978). *Lucina* that dominated at Herring Bay in 1989 (61% of faunal abundance) were mainly older than one year; many noted were more than three years old.

Similar surveys at Herring Bay in 1990 revealed fewer dead animals than in 1989. In 90m² surveys conducted in late May and early October, we saw one dead *Pycnopodia* (spring), one dead cod and three dead worms (fall) among scattered patches of *Beggiatoa*. More extensive visual searches in fall 1990 revealed some dead fishes, but there were no concentrated "dead zone" pockets as observed in 1989.

However, the infaunal community was obviously disturbed by the spring and fall of 1990. Diversity and number of taxa were extremely low in both surveys, <0.1 and <8 taxa (mainly families), respectively. Furthermore, *Lucina* was absent and the community was almost totally dominated by a single polychaete, *Nephtys cornuta*. Dissolved oxygen values in the water adjacent to the bottom averaged 5.4 mg/l in May, but were near zero in October.

Lizarraga-Partida (1974) reported *Nephtys cornuta* in semi-polluted substrates in Ensenada Bay, Mexico, in areas enriched with organic material derived from sewage-fish waste. Pearson and Rosenberg (1978) give several other examples of *Nephtys* appearing in organically enriched and polluted areas, often low in dissolved oxygen. Busdosh (1978) found *Nephtys* only in association with oiled substrate; it alone preferred oiled to clean sediment. Although *Nephtys* is mainly a predator, it also deposit feeds and thus can utilize the high organic loads associated with the decay of dead organisms.

By mid August 1991, the community in Herring Bay demonstrated dramatic signs of recovery. Diver surveys revealed no dead organisms, although *Beggiatoa* was still evident on the bottom. Almost all infaunal community parameters had recovered to or near levels observed in 1989. *Nephtys cornuta* still dominated, however, *Lucina tenuisculpta* was now present again, but in low density. The dissolved oxygen during the August sampling, one month earlier than 1989 and 1990 fall samplings, averaged 9.7 mg/l.

One group of organisms that was present in 1991 in moderate density was burrowing amphipods (132/m²). Representatives of this group included the fami-

lies Ischyroceridae, Isaeidae, Dexaminidae, Phoxocephalidae, and Lysianassidae. The low density of amphipods at Herring Bay in 1989 (16/m²) and 1990 (absent), presumably was the direct result of toxic effects of petroleum hydrocarbons. Benthic amphipods are notoriously sensitive to petroleum hydrocarbons, and massive declines in ampeliscid amphipods were observed following the *Amoco Cadiz* oil spill (Cabioch *et al.*, 1978; Chasse, 1978). The reoccurrence of amphipods on those previously contaminated sediments began, for some species, in one to two years (Dauvin, 1982). The density of amphipods had dramatically increased in Herring Bay by 1991, just two years since being oiled.

Hydrocarbon compounds were identified from surficial sediments from Herring Bay fjord. A gradual decline in each hydrocarbon compound was noted over the three-year period, e.g., total hydrocarbons decreased from a mean of 69.0 µg/g in 1989 to 3.7 µg/g in 1991; total polynuclear aromatic hydrocarbons decreased from a mean of 1.2 µg/g in 1989 to 0.06 µg/g in 1991; total dibenzothiophenes decreased from a mean of 0.222 µg/g in 1989 to 0.006 µg/g in 1991; total alkanes decreased from a mean of 2.0 µg/g in 1989 to 0.67 µg/g in 1991; and total naphthalenes decreased from a mean of 0.022 µg/g in 1989 to 0.012 µg/g in 1991. Aromatic petroleum hydrocarbons, including naphthalenes and phenanthrenes, have been reported to persist in sediments for periods in excess of six years, although in reduced concentrations (Neff and Anderson, 1981).

All four of the other silled fjord sites (two oiled and two unoled) that were examined during late September 1990

were stressed as indicated by the low number of taxa (4-13), low Shannon diversity indices ($H' < 0.1-1.3$), and high Simpson dominance indices ($D = 0.3-0.9$). All sites were dominated by *Nephtys cornuta*. All sites also had bottom-water dissolved oxygen values of <1 mg/l and the sulfur-dependent bacteria, *Beggiatoa* present. Dead organisms, such as juvenile Pacific herring and unidentified worms, were found in both oiled and unoled sites. Other dead organisms on oiled sites included terebellid polychaetes, naticid snails, and brittle stars. At one unoled site (Humpback Cove), where oxygen values were near zero, the sulfur odor could be detected by the divers at depth.

Our data from Herring Bay tends to support the notion that oil contributed to the "dead zone" observed in 1989, and the infaunal reduction in 1990. That extensive "die off" of organisms occurred in 1989, concomitantly with high concentrations of hydrocarbons in the sediment, underscore an oiling effect. This environment must have been particularly stressed, since pelagic organisms also died.

The effects of oil on subtidal benthic communities at similar or deeper depths have been observed in other studies (e.g., Cabioch *et al.*, 1978; Hyland *et al.*, 1989). Among the possible causes of oil-related effects are chemical toxicity of aromatic derivatives; asphyxiation or entanglement due to direct physical coating; and a variety of reproductive, behavioral, and other sublethal disorders leading ultimately to long-term population changes. In the case of the *Exxon Valdez* spill, there also is the possibility of indirect effects of bioremediation and other clean-up efforts. Such activities were observed, for example, at the heavily-oiled Herring

Bay site.

The organisms that inhabit silled fjords, like the fjordic portion of Herring Bay, are periodically subjected to stress caused by, or confounded by, hypoxic or anoxic conditions on the sea floor related to poor water exchange and natural cycles of organic enrichment. Similar effects on the benthos from low-oxygen stress and organic enrichment have been seen in Scandinavian and Scottish fjords (review by Pearson, 1980) and documented as the "August Effect" in New England estuaries (Rhoads and Germano, 1982).

In conclusion, it is likely that the observations made in the fjordic portion of Herring Bay during 1989 and 1990 were the result of exposure to *Exxon Valdez* crude oil, in conjunction with hypoxic/anoxic conditions. However, further benthic studies should be conducted in Prince William Sound fjords to provide information critical in distinguishing possible oil impacts from the natural phenomena of hypoxia and anoxia.

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The Effects of the Exxon Valdez Oil Spill on Epibenthic Invertebrates in the Shallow Subtidal

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Studies were conducted in 1990 and 1991 to examine the effects of the Exxon Valdez oil spill on the shallow subtidal community in Prince William Sound. Here we will present results from a stratified random sampling program that compared population densities of numerically dominant species of large epibenthic invertebrates at oiled and control sites.

Sampling in 1990 was conducted within 4 different habitats within the Sound: Eelgrass habitats in shallow protected bays, *Laminaria/Agarum* habitats in sheltered bays (hereafter referred to as bays), *Laminaria/Agarum* habitats on exposed points (hereafter referred to as points), and in *Nereocystis* habitats on exposed coastlines. Within each habitat we sampled at 2 to 4 pairs of oiled and control sites. The oiled sites were selected from areas that had adjacent shorelines that were moderately to heavily oiled during the fall of 1989. Control sites were selected that matched the oiled site with respect to physiographic factors (e.g. exposure, slope, substrate type) but that did not have oil present on adjacent shorelines in fall 1989. We sampled within one depth stratum in eelgrass and *Nereocystis* habitats, and within two depth strata at bay and point habitats. Divers counted the number of large benthic invertebrates along three randomly placed 30 m x 2 m transects within each site and depth stratum.

In 1991, we sampled only in eelgrass beds and within the shallower depth stratum of bays. The sites sampled within

these habitats were the same sites sampled in 1990, except that 1 additional pair of sites was sampled within the eelgrass habitat.

We will focus here on the patterns of abundance of the dominant epibenthic invertebrates within these habitats. The dominants were the helmet crab (*Telmessus cheiragonus*) and two species of starfish: the leather star (*Dermasterias imbricata*) and the sunflower seastar (*Pycnopodia helianthoides*). Other species of starfish were present (e.g. *Evasterias troschelii*, *Henricia leviuscula*, and *Orthasterias koehlerii*) but these were less abundant than *Dermasterias* or *Pycnopodia*, and did not occur in all habitats. Evidence of the impacts of oiling is based primarily on the data gathered in 1990. Data collected in 1991 are used to indicate recovery of affected populations, or lack thereof.

Populations of both leather stars and helmet crabs appeared to be adversely affected by oiling and/or associated cleanup activities. In 1990, the mean density of *Dermasterias* was significantly greater ($P < 0.05$) at control sites than at oiled sites in all habitats combined. On average, the starfish were about twice as abundant at oiled sites than at controls. Similarly, the helmet crab, *Telmessus cheiragonus*, was more abundant at control sites relative to oiled sites ($P < 0.01$ for all habitats combined). The average population density of *Telmessus* was about 5 times as high at control sites than at oiled sites in 1990.

There were no apparent adverse impacts of oil on *Pycnopodia*. Population densities of adult *Pycnopodia* were similar at oiled and control sites in both 1990 and 1991.

There was little indication of recovery of the *Dermasterias* population, as significantly more leather stars were found at control sites relative to oiled sites again in 1991 ($P < 0.01$). There was a slight increase in the mean abundance of *Dermasterias* from 1990 to 1991, but the means for each year did not differ significantly ($P = 0.09$ and $P = 0.85$ in eelgrass and bay habitats respectively).

Unlike *Dermasterias*, population densities of *Telmessus* showed some signs of recovery in 1991. There were no significant differences in *Telmessus* density among oiled and control sites in either eelgrass ($P = 0.23$) or bay habitats ($P = 0.27$) in 1991. However, the parity noted among oiled and control sites in 1991 resulted largely from a decrease in population density at the control sites rather than an increase at the oiled sites, and may have resulted from immigration of crabs from control to oiled areas following a reduction in hydrocarbon levels in 1991.

Also, in 1991, we noted large numbers of newly settled *Pycnopodia*, especially in the eelgrass habitats. Densities of young of the year *Pycnopodia* averaged over 30 per 100 m² in eelgrass habitats and reached densities of more than 200 per 100 m² at one site on Naked Island. Significantly more young of the year *Pycnopodia* were observed at oiled than control sites in the bay habitats ($P < 0.01$), suggesting a possible increase in *Pycnopodia* recruitment at oiled sites. However, no significant differences were observed among oiled and control sites in eelgrass habitats ($P = 0.12$).

We can only speculate as to the causes for the observed effects of oiling. However, we suspect that *Dermasterias* populations at oiled sites may have been reduced as a result of acute oil toxicity or by the effects of cleanup activities (especially steam cleaning). *Dermasterias* is most abundant in the shallow subtidal along rocky shorelines. They often migrate into the shallow intertidal where they feed during high tides, are exposed briefly as the tide retreats, and then drop off of rather steep rocky shores into the water. This behavior may have made these starfish especially vulnerable to oiling and to shoreline cleanup activities.

Previous studies of the toxicity of oil to asteroids suggest that low concentrations of oil may impair the feeding ability of starfish (O'Clair and Rice, 1985). However, only very high concentrations of oil are lethal to adult asteroids (reviewed in O'Clair and Rice, 1985). As a result, we suspect that losses of *Dermasterias* at oiled sites were more likely the result of cleanup activities. *Pycnopodia* can also be found in the intertidal, but unlike *Dermasterias*, seldom tend to migrate from the subtidal into the intertidal. As a result they may have been spared from direct impacts of oil and shoreline cleanup.

We suspect that lower densities of *Telmessus* at oiled sites may have resulted from losses due to the toxicity of oil. Crustaceans are especially sensitive to oil (Capuzzo, 1987) and may have been killed by high concentrations of oil associated with the spill. Alternatively, these crabs may have been able to flee the oiled areas, and migrate back to oiled sites in 1991 after hydrocarbon levels had declined.

The possible increase recruitment of *Pycnopodia* at oiled sites in 1991 may be an indirect effect of oiling. We have

noted increased recruitment at oiled sites for a number of species including *Pycnopodia*, several species of fish (Pacific cod and cottids) and small mussels (*Musculus* sp.), and we suspect that these differences may result from enrichment of waters near sites of extensive bioremediation. However, an alternative hypothesis is that oiled sites tend to be areas where prevailing currents and winds concentrated both oil and larvae.

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The Effects of the Exxon Valdez Oil Spill on Eelgrass and Subtidal Algae

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Studies were conducted in 1990 and 1991 to examine the effects of the Exxon Valdez oil spill on eelgrass (*Zostera marina*) and dominant subtidal algae in shallow subtidal habitats in Prince William Sound. Sampling in 1990 was conducted within 4 different habitats within the Sound: Eelgrass habitats in shallow protected bays, *Laminaria/Agarum* habitats in sheltered bays (hereafter referred to as bays), *Laminaria/Agarum* habitats on exposed points (hereafter referred to as points), and in *Nereocystis* habitats on exposed coastlines.

Within each habitat we sampled at two to four pairs of oiled and control sites per habitat. The oiled sites were selected from areas that had adjacent shorelines that were moderately to heavily oiled during the Fall of 1989. Control sites were selected that matched the oiled site with respect to physiographic factors (eg. exposure, slope, substrate type) but that did not have oil present on adjacent shorelines in Fall 1989. We sampled within one depth stratum in eelgrass and *Nereocystis* habitats, and within two depth strata at bay and point habitats.

For eelgrass, we examined a variety of population parameters including percent cover, the density of turions (uprights protruding from the substrate) and the density of flowers. Divers counted the number of turions and estimated percent cover in each of 4 - 0.25 m² quad-

rats along each of three randomly placed 30 m x 2 m transects within each site.

In the bay, point and *Nereocystis* habitats, divers estimated the percent cover and counted the number of dominant understory algae (Eg. *Agarum cribrosum*, and *Laminaria spp.*) in each depth stratum. In addition, divers estimated the abundance of larger *Nereocystis* plants within a 2-m-wide band along each transect and measured the diameter of the stipe of the first 20 plants encountered. Preliminary studies indicated that stipe diameter was a good indicator of length and weight of each plant.

In 1991, we sampled only in eelgrass beds. The sites sampled within these habitats were the same sites sampled in 1990, except that 1 additional pair of sites was sampled.

The density of eelgrass turions was approximately 30% greater at control sites than at oiled sites in 1990 ($P=0.08$), and there were significantly fewer flowering plants at oiled sites ($P=0.06$). By 1991, eelgrass had apparently recovered as there were no differences noted among sites with respect to either turion density ($P=0.52$) or flower density ($P=0.60$).

The dominant plants in bay habitats were the stipate kelps *Agarum cribrosum* and *Laminaria saccharina*. The density and percent cover of *Laminaria spp.* (the vast majority of which were *L. saccharina*) were greater at the oiled sites relative to the control sites, in both deep and shal-

low depth strata ($P < 0.05$ in all cases). *Laminaria* represented about 45%, on average of the total algal cover at oiled sites, but only 13% of the total cover at control sites.

The density of *Agarum* did not differ among sites ($P = 0.24$ and $P = 0.31$ at shallow and deep strata respectively). However, there were observable differences with regard to size distributions of *Agarum*. There tended to be proportionally more small plants, and proportionally fewer large plants at the oiled sites, especially in the shallower depth stratum. The size distributions differed significantly at the shallower depth stratum ($P < 0.01$) and were nearly significant ($P = 0.15$) in the deeper strata.

Points around the islands of the Knight Island group tended to have slightly higher algal diversity than the bays, but were still dominated by *Agarum cribrosum* and *Laminaria saccharina*. There were generally higher densities of *Agarum* at the oiled than at control sites ($P = 0.07$ shallow and $P = 0.02$ deep). These differences were primarily attributable to significantly greater numbers of small *Agarum* (<10 cm in height) at the oiled sites ($P = 0.03$ shallow and $P < 0.01$ deep). Also, the size distributions of *Agarum* (for plants larger than 10 cm) revealed a pattern similar to that observed in the bays, with proportionally fewer large individuals and more smaller plants at the oiled sites, especially in the shallower depth stratum. However, the size distributions did not differ significantly ($P = 0.15$ shallow and $P = 0.32$ deep).

Nereocystis habitats had a canopy of *Nereocystis leutkeana* with a diverse understory consisting of primarily of *Agarum cribrosum*, *Pleurophycus gardneri*, and 3 *Laminaria* species (*L. saccharina*, *L. groenlandica*, and *L. yezoensis*). *Nereocystis*

density was more than 7 times greater at the oiled sites relative to the control ($P < 0.01$). Also, there were proportionally more small plants and fewer large plants at the oiled sites ($P = 0.10$).

Eelgrass appears to have been adversely affected by the spill as evidenced by the lower densities of turions and flowers at the oiled sites. The evidence of an effect is strongly supported by similar evidence presented in a second, independent study. An evaluation of the effects of the Exxon Valdez oil spill on eelgrass in Prince William Sound by Teas et al. (1991) also demonstrated that there was a reduced density of flowering stalks of eelgrass at oiled sites.

In each of the three habitats on hard substrate (*Nereocystis*, *Laminaria*/*Agarum* in bays, and *Laminaria*/*Agarum* on points) we observed differences in the size distribution of the dominant alga at oiled and control sites. In all cases, there were proportionally more small algae (*Nereocystis* and *Agarum* in the *Nereocystis* habitat and *Agarum* at the other two habitats) at the oiled sites. In addition, we observed higher mean densities of one of the dominant algal species at oiled sites in each habitat: *Nereocystis* in the *Nereocystis* habitat, *Laminaria saccharina* in the bays, and small *Agarum cribrosum* at points.

We interpret these differences in size distribution and density as recovery of the algal community following a loss of algae in 1989. We suspect that the spill was responsible for the loss of algae, either as a direct effect of oil or from cleanup activities, and that the free space created was quickly colonized by new recruits in 1990. An alternative hypothesis is that the differences in size distribution were the result of slower growth of algae at oiled sites.

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The Effects of the Exxon Valdez Oil Spill on Infaunal Invertebrates in the Eelgrass Habitat of Prince William Sound

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Sampling was conducted in 1990 and 1991 to assess the impacts of the Exxon Valdez oil spill on infaunal invertebrates within and adjacent to shallow (<20 m) subtidal eelgrass (*Zostera*) beds in western Prince William Sound. We present results from a stratified random sampling program that compared measures of diversity, abundance and biomass of mainly infaunal invertebrates at paired oiled and control (unoiiled) sites.

The oiled and unoiiled site pairs for both years were Bay of Isles-Drier Bay, Herring Bay-Lower Herring Bay, Sleepy Bay-Moose Lips Bay, and Clammy Bay-Puffin Bay, respectively. The oiled sites were selected from areas that had adjacent shorelines moderately to heavily oiled during the fall of 1989. Control sites were selected that matched the oiled sites with respect to physiographic factors (e.g., exposure, slope, substrate type), but that did not have oil present on adjacent shorelines in fall 1989. Two depth strata (6-20 m and within the eelgrass bed [< 3 m]) were sampled at each site. Three stations were established within each depth stratum and two 0.1 m² benthic samples were collected from each station with a diver-operated suction dredge. Sediment samples were concurrently collected for grain size and hydrocarbon analyses.

We tested for differences among oiled and unoiiled sites using a randomization procedure (Manley, 1991). This procedure is briefly summarized as follows. (1) The blocked analysis of variance was

performed. Station means were used as replicates, and replicates were blocked by oiled/unoiiled pair. A sum of squares (SS) was produced for each factor. (2) Next, using the original data set, we randomly reassigned values for oil code to each station value. The ANOVA was then rerun on this new data set. (3) Step 2 was repeated 1000 times. (4) The SS from the ANOVA of the original data set was compared with SS of the 1000 randomly drawn data sets. The proportion of instances in which the SS for the randomly drawn data exceeded the SS for the original data was recorded. This value is the significance level of the test. The significance level is interpreted in the same manner as for parametric procedures. Analyses were conducted on diversity, dominance, species richness, total abundance, total biomass, total taxa, and the highest 15 ranking taxa (typically families) for abundance and biomass within each depth stratum. Separate analyses were performed that examined differences among oiled and unoiiled sites within each year (1990 and 1991). In addition, 2-way analyses were performed that examined differences among oil and unoiiled sites, differences among years, and their interaction.

In 1990, the health of the benthic community at 6-20 m depths, on mud/sand substrates adjacent to the eelgrass bed, was generally better at unoiiled than at oiled sites. Conversely, the benthic community within the eelgrass bed was typically more robust at oiled sites. Further-

more, infaunal invertebrates were generally less abundant at the oiled sites, while epifauna tended to be more abundant at the oiled sites. In deeper waters adjacent to the bed, the biomass of three (Caecidae, Lepetidae, Veneridae) of the 15 dominant taxa was greater ($p < 0.05$) at the unoiled sites, while only one family (Maldanidae) was more abundant ($p = 0.02$) at the oiled sites. Within the eelgrass bed, several epifaunal families (e.g., spirorbids, spionids, mytilids, lacunids) were more abundant (or had greater biomass) at the oiled sites. Two dominant families (trochid snails and phoxocephalid amphipods) were more abundant ($p < 0.01$ for each family) at the unoiled sites.

The differences among oiled and unoiled sites appear related to oiling or cleanup activities. A preliminary analysis of hydrocarbon data suggest that in 1990, there were generally higher concentrations of hydrocarbons in the sediments from oiled sites.

There were indications of recovery of the epifaunal community in 1991. There were no differences among oiled and unoiled sites with respect to any of the diversity measures, or with respect to total biomass or total abundance. However, there were still four of the 15 dominant families that had greater biomass at the unoiled sites relative to the oiled sites.

Comparisons between years (oiled and unoiled sites combined) indicated that there were significantly greater abundance ($p < 0.01$) and a greater number of taxa ($p < 0.05$) in 1991 relative to 1990, at both depth strata. Seven families were significantly ($p < 0.1$) more abundant in 1991 vs. 1990, and ten families had greater ($p < 0.1$) biomass in 1991. None of the families displayed a significant decrease

in abundance or biomass in 1991.

Comparisons between oiled vs. unoiled sites (1990 and 1991 combined) indicated that there were significantly ($p < 0.1$) greater diversity and total biomass at unoiled sites relative to oiled sites, at both depth strata.

The increase in the abundance of infauna in 1991 may have been related to a recovery from oiling effects at both oiled and unoiled sites. While the hydrocarbon concentrations were generally higher at oiled than unoiled sites in 1990, there was a substantial decrease in concentrations of hydrocarbons present at both oiled and unoiled sites in 1991.

Perhaps the greatest single indication of initial oil effects, followed by recovery, was the recolonization of oiled sites by sensitive burrowing amphipods. In 1990, at 6-20 m depths, the abundance of all amphipods was significantly ($p < 0.01$) greater at unoiled sites (unoiled mean = 47 individuals/0.1 m²; oiled mean = 19/0.1 m²). However, at this depth in 1991, no difference ($p > 0.1$) in total amphipod abundance or biomass was detected between oiled and unoiled sites. It is likely that the low density of amphipods at oiled sites in 1990 was the direct result of toxic effects of petroleum hydrocarbons. Benthic amphipods are notoriously sensitive to petroleum hydrocarbons, and massive declines in ampeliscid amphipods were observed following the *Amoco Cadiz* oil spill (Cabioch *et al.*, 1978; Chasse, 1978). The reoccurrence of amphipods on those previously contaminated sediments began for some species in 1 to 2 years (Dauvin, 1982). Within Prince William Sound, these crustaceans had reoccupied the oiled sites in significant densities by 1991, just 2 years since being oiled.

Although the negative effects were

most apparent in this study, some increases in abundance or biomass were observed at oiled sites. The increases at the oiled sites were primarily attributable to the small epifaunal suspension feeding spirorbid and spionid polychaetes and mytilid mussels (*Musculus* spp.), and to a lesser extent by the surface deposit-feeding polychaetes, Nepthyidae and Maldanidae. This may have been an indirect affect of oiling or cleanup, as the result of a reduction in predator abundance. However, it is possible that oiled sites tend to be areas where prevailing currents and winds concentrate both oil and larvae.

The most recent differences among oiled and unoiled sites seem to be indicative of advances in recovery of the community. We would expect the recovery to be more rapid at shallow depths. These are higher energy areas where sediments are more frequently reworked by waves, and where oil is less likely to persist. The recovery at the oiled sites may have been ameliorated by bioremediation activities that led to local enrichment of microbial and plankton communities.

Results from previous oil spills indicate that benthic communities generally represent good *in situ* monitors for measuring effects of oil fluxing to the bottom (e.g., Kineman *et al.*, 1980) and that moderate amounts of oil in sediment cause

impacts comparable to those we observed.

Most post-spill environmental studies elsewhere have been 3 to 5 years in duration. To date, only 2 years of monitoring has occurred, with no sampling in 1992. Collectively, our findings suggest an oiling effect on benthic invertebrates, since the components tested over the 2-year period showed that unoiled sites had the more robust community and that the community was recovering over the short period studied. However, conclusions on the effects from the spill cannot be made until the hydrocarbon data are fully analyzed.

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Assessment of Injury to Pink Salmon Eggs and Fry

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This study is part of an integrated group of Natural Resources Damage Assessment Fish/Shellfish Studies (NRDAF/S) conducted to quantify damage to pink salmon *Oncorhynchus gorbuscha* as a result of the Exxon Valdez oil spill. Each study attempted to determine the injury to salmon at different stages of the life cycle. Wild pink salmon play a major role in the Prince William Sound ecosystem. Salmon are prey to a variety of terrestrial and marine mammals and birds, while also providing a pathway for nutrient transfer from marine to near-shore and terrestrial ecosystems. Wild pink salmon also contribute to the region's commercial fisheries.

Up to 75% of the wild pink salmon which spawn in Prince William Sound use intertidal areas (Helle et al. 1964). These areas are highly susceptible to contamination from marine oil spills. Moles et al. (1987) and Rice et al. (1975) found that pink salmon eggs and pre-emergent fry were adversely affected by exposure to crude oil and that the effect was most acute in intertidal environments. The 24 March 1989 spill from the Exxon Valdez occurred just prior to the spring migration of salmon fry and contaminated many intertidal spawning areas in central and southwest Prince William Sound.

This study evaluated (1) the immediate effects of oil exposure on pre-emergent pink salmon numbers in the spring of 1989, (2) the effect of intertidal oil exposure on pink salmon egg mortality, and (3) the effect of intertidal oil exposure on pink salmon egg to pre-emergent

fry survival. Samples were also collected for histopathological and mixed-function oxidase analysis. This project concentrated on southwestern Prince William Sound although streams from Montague Island and eastern Prince William Sound were sampled to provide a broader perspective.

Study streams were selected using the following criteria: (1) adult salmon returns were expected to be large enough to provide a high probability of success in egg and fry sampling, (2) egg and fry sampling had been done in past years, and (3) streams which had low to no oil impact (controls) were selected near high oil impact streams. Pink salmon fry remain in the area in the stream where they were deposited as eggs. This trait allowed oiled and control sites to be located in close proximity to each other, thus reducing any geographical effect on the findings.

Forty-eight streams were sampled for pre-emergent fry in 1990, 1991, and 1992. These included 25 streams historically sampled to forecast adult pink salmon returns and 23 additional streams from the oil impact area. Thirty-one streams were sampled for pink salmon egg mortality in 1989, 1990, and 1991. The streams sampled for egg mortality were included in the group of streams sampled for pre-emergent fry.

The methods used for both egg and pre-emergent fry sampling were similar to those described by Pirtle and McCurdy (1977). Sampling was stratified by tide zone to control for possible differences in

egg mortality or overwinter survival due to salinity, temperature, predation, oil, or a combination of these factors. Four zones, three intertidal and one above tidal inundation were sampled, whenever possible, for each stream: 1.8 - 2.4 m, 2.4 - 3.0 m, and 3.0 - 3.7 m above mean low water, and upstream of mean high tide (3.7 m). Zone boundaries were established with a surveyor's level and stadia rod and staked prior to sampling. No sampling was done below the 1.8 - 2.4 m zone as survival was expected to be low (Helle et al., 1964). Upstream sample areas were often within the reach of extreme high tides (3.7 - 4.6 m) since ice and snow often limit the extent of upstream sampling.

Separate linear transects were established within each zone for egg and pre-emergent fry surveys. Although most transects were 30.5 m long, some were shorter due to steep stream gradients. Transects were placed in riffle areas where spawning was observed during escapement surveys conducted by NRDA F/S Study 1. Transects ran diagonally across the stream: fry survey transects started downstream against the right bank and moved upstream to the left bank, while egg survey transects started downstream against the left bank and moved upstream to the right bank. This placement of egg and fry transects reduced sampling overlap and the influence of fall egg sampling on spring fry abundance.

Fourteen circular digs, each 0.186 m², were systematically made along each transect. The number of digs was a compromise between reducing variance and the practicality of conducting the study. Fewer digs were completed in narrow stream channels to avoid excessive sampling of the stream.

Stream oil exposure classifications were based on visual observations (NRDA F/S Studies 1 and 2) and hydrocarbon content of 1989 mussel tissue (*Mytilus* sp.) samples (NRDA F/S Study 1). Hydrocarbon analysis of mussel tissue and mixed-function oxidase analysis of pre-emergent fry generally agreed with visual observations of stream oil contamination. Histopathological analysis failed to detect lesions in pre-emergent fry, although results from another study (Fink, 1992) indicate the fry may have been collected too early in their life to have developed lesions.

Since the annual pre-emergent pink salmon fry density survey conducted by the Alaska Department of Fish and Game, Division of Commercial Fisheries, was underway at the time of the spill, many streams were sampled for pre-emergent fry density prior to or immediately after oil exposure. An additional session of sampling was also done approximately two weeks after the spill. This second survey allowed some streams examined during the first sampling session to be examined for immediate effects of oil contamination.

Few dead pink salmon fry were found either prior to or shortly after oil exposure. Only nine of the 52 transects examined contained more than five dead fry. No increase in fry mortality was detected between the first and second samplings, although only three of the 14 streams examined were oiled. Likewise, no difference in fry density was detected between the first and second sampling.

Egg mortality was significantly greater in oiled streams in 1989, 1990, and 1991. We believe these differences indicate an effect due to oil exposure. The 1989 investigation detected a statistically significant difference in egg mor-

tality ($p=0.0001$) between oiled and control streams. Examination of stream zone contrasts indicated that egg mortalities were greater in oiled streams and that statistical differences were due to elevated egg mortality in the intertidal zones. Mean mortalities for the oiled and control streams were 0.174 and 0.104, respectively.

The 1990 egg mortality study also showed a statistically significant difference ($p=0.0008$) between oiled and control streams. Again, examination of stream zone contrasts indicated greater mortalities in oiled streams with the statistical difference confined to the upper intertidal zone. Mean egg mortalities for the oiled and control streams were 0.295 and 0.195, respectively.

Egg mortality results were consistent with perceived oil contamination: among oiled streams, all intertidal zones were contaminated in 1989 whereas in 1990 oil remained only in the upper intertidal zone.

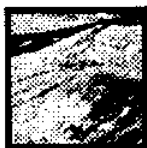
The 1991 evaluation demonstrated very significant egg mortality differences between oiled and control streams ($p=0.0001$). Inspection of stream zone contrasts indicated that egg mortalities in all zones were greater for the oiled streams. Mean mortalities for the oiled and control streams were 0.433 and 0.221, respectively. This finding was unexpected and at this time remains unexplained. We have hypothesized that the continuing and increased mortality is the result of genetic damage sustained by the eggs and alevins which incubated in oiled gravel during the fall of 1989 and spring of 1990. We are presently evaluat-

ing this hypothesis through a series of controlled rearing experiments.

No significant difference in egg-to-fry survival was detected between oiled and control streams for 1989 to 1990, 1990 to 1991, or 1991 to 1992. We feel these results were due to insufficient power in the sampling design or sampling levels to detect differences rather than a true lack of change.

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Cytochrome P450 Induction and Histopathology in Pre-emergent Pink Salmon From Oiled Streams in Prince William Sound, Alaska

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The 24 March 1989 *Exxon Valdez* spill in Prince William Sound and the Gulf of Alaska oiled a minimum of 213 intertidal pink salmon spawning streams. Some stream sediments remained contaminated through spring 1992. Staff from the Alaska Department of Fish and Game, Habitat Division, developed spill response recommendations and monitored the clean-up of oiled anadromous streams. After initial spill clean-up in summer 1989, the ADF&G began studies (Fink, 1992; Wiedmer, 1992) to determine if pre-emergent pink salmon eggs and fry were impacted by remaining petroleum residues. In December 1989; May, June, and September 1990; and March and May 1991, pre-emergent pink salmon eggs and fry were collected from four oiled and five control sites in Prince William Sound. A total of 23 fry samples consisting of 6 individuals each and 10 egg samples consisting of at least 6 eggs each were collected from mid to upper intertidal zones at oiled and control sites. Contaminated spawning sediments were analyzed utilizing gas chromatography with flame ionization detection (GC/FID) and gas chromatography with mass spectrometry (GC/MS).

To assess the continuing exposure of eggs and fry to hydrocarbon-contaminated spawning gravels, we evaluated the induction of cytochrome P450. Cytochrome P450 proteins are monooxygenase (MO) enzymes which mediate the metabolism of xenobiotics such as polycyclic aromatic hydrocarbons (PAHs).

Elevated levels of a specific P450 form, P450IA, have been found to indicate exposure to environmental pollutants (Stegeman and Lech, 1991). The P450IA content was evaluated by immunohistochemical analysis using monoclonal antibody 1-12-3 (Park et al., 1986; Smolowitz et al., 1991) specific for P450IA forms. Induction was ranked as negative, very mild, mild, moderate, or strong. To determine whether there were histopathological effects, Drs. Hinton and Marty, University of California, Davis, examined larval sections for the presence and severity of lesions. Lesions were ranked as none, mild, moderate, or severe.

Fry samples were stratified by time and pooled for each stream. The level of biological impact for each stream and time strata was estimated by the median of cytochrome P450IA induction and lesion scores. The Mann/Whitney (Wilcoxon) two-sample test (Conover, 1980) was used to test for statistical differences between oiled and control streams.

Cytochrome P450IA induction was observed in endothelial and epithelial cells of several organs of post-hatch larvae (13 of 16 samples) from oiled streams, and appeared to be independent of developmental stage. Organs exhibiting positive immunochemical staining included kidney, gill, liver, intestine, heart, brain, yolk sac, skin, peritoneal connective tissue, and pharyngeal epithelium. In 1990, induction in fry samples ranged

from negative to strong, while induction in 1991 ranged from negative to moderate. All seven fry samples from control streams were negative for induction. P450IA induction was not observed in pre-hatch egg samples, regardless of hydrocarbon exposure (December 1989 and September 1990).

P450IA content was significantly elevated in oiled streams in May and June, 1990 ($P < 0.067$) and March 1991 ($P < 0.10$). No statistical tests were performed on fry samples collected during December 1989 due to small sample size; however, examination of staining results indicated a difference may be present in P450IA expression between oiled and unoled sites. No significant induction was observed in May 1991 ($P < 0.20$). There was no significant change in induction intensity between 1990 and 1991.

Histopathological lesions were found in fry from six of 16 samples (35 of 93 fry, 38%) collected within the oiled zone of impacted streams in 1990-91. Lesions were found in fry from three of seven control samples (7 of 42 fry, 17%). While small sample size prevented meaningful statistical analysis, examination of median scores suggests an increase in lesions among fish collected from oiled streams in 1990, but not in 1991. Epidermal atrophy (EA) was the most frequently observed lesion. Other lesions observed included myofiber degeneration and necrosis (MDN), and individual hepatocellular degeneration and necrosis (IHN). Both MDN and IHN were absent in 1991 samples.

The occurrence of lesions appeared to correlate with the absorption of the yolk sac. Lesions were not found in post-hatch samples collected from oiled streams in December 1989, or in March 1991 (early life stages). However, lesions

were found in samples collected in May/June 1990, and May 1991 (near the time of emergence).

Analyses of sediments from oiled sites indicated that polynuclear aromatic hydrocarbon constituents of *Exxon Valdez* crude contaminated these sites (GERG, 1990).

Immunohistochemical staining results indicate that pre-emergent pink salmon fry from oiled sites in Prince William Sound were responding to chronic exposure to PAHs by activation of the P450IA genes. While induction of P450IA can be caused by various contaminants, evidence from this study suggests that PAHs caused the observed induction. As a result of MO-mediated metabolism, toxicity of PAHs greatly increases (Wood et al., 1976). The products of MO-mediated metabolism bind to the cellular macromolecules RNA (Blobstein, et al., 1976) and DNA (Sims et al., 1974). The degree of metabolite binding is positively correlated to carcinogenic (Brookes and Lawley, 1964) and mutagenic (Jerina and Daly, 1974; Stegeman and Lech, 1991) potential. These observations indicate that elevation of cytochrome P450IA in pre-emergent pink salmon from Prince William Sound oiled streams may create an increased potential for mutagenesis and carcinogenesis.

Fish exposed to xenobiotics have demonstrated an inverse correlation between MO-activity and androgen, estrogen, and corticoid concentrations (Sivirajah et al., 1977), and testicular development (Truscott et al., 1983). Spies et al. (1988), found an inverse correlation between MO activity and parameters of female reproductive success (e.g., decreased proportion of viable eggs, decreased fertilization success, decreased embryological

success). These observations suggest that the reproductive potential of salmon developing in contaminated sediments may be diminished. The fish of this study were exposed to oil during their most dynamic developmental period; a period of rapid cellular differentiation and organogenesis. Pink salmon are semelparous, they reproduce but once. Factors with deleterious effects on the initial organogenesis of gonadal tissue will directly affect life time reproductive success. Changes in hormonal concentrations during the early life stages of oil-exposed fish may decrease parameters of reproductive potential such as egg number, size, and viability.

Histopathological results suggest that petroleum exposure may have caused an increase in lesion occurrence. Lesions appeared to be more severe and of greater frequency in 1990 samples as compared to 1991 samples. This might be expected if hydrocarbon concentrations within contaminated spawning substrates were reduced as a result of natural cleaning by winter storms, by other abiotic and biotic factors, and by stream treatments in the summer of 1990 at the four oiled streams.

In this study, while cytochrome P450IA induction was found in fry collected from oiled streams in December through early June, the occurrence of lesions in fry appeared to be developmental-dependent. Sac fry with abundant yolk did not exhibit lesions, while fry with minimal yolk reserves had higher probability of developing lesions, particularly epidermal atrophy. The skin is the first line of defense against pathogens and is also important in osmoregulation. Fry undergo physiological changes upon emergence, and subsequent acclimation into saltwater. Additional physiological stresses may affect

an individual fry's ability to avoid predation, thereby affecting survival.

The current study demonstrates that pre-emergent pink salmon fry in some heavily impacted streams incorporated Exxon Valdez petroleum into various tissues more than two years after the initial spill and that the hydrocarbons and their metabolites induced detectable physiological changes. These results can provide valuable links between the Exxon Valdez spill and any biological injuries in pink salmon observed in other studies.

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Application of Half Length Coded-Wire Tags to Wild and Hatchery Pink Salmon Fry in Prince William Sound, 1989-1991.

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Accurate estimates of wild and hatchery pink salmon *Oncorhynchus gorbusha* contributions to the commercial catch are essential to evaluate stock specific damage from the T/V Exxon Valdez oil spill. A mark is required for investigators to distinguish specific stocks during juvenile and adult life stages in order to quantify stock-specific differences in growth rates and fry to adult survival rates which may have been caused by oil damage. Coded-wire tags applied to pink salmon fry released from four Prince William Sound Hatcheries in 1989, 1990, and 1991 and to wild fry from three oiled and three unoled streams in 1990 and 1991 provided such a mark.

Each year more than 500 million fry were enumerated and released from four hatcheries in Prince William Sound. Half-length coded wire tags were applied to approximately one out of every 560 fry released in 1989, 1990, and 1991. Approximately 2.2 million wild fry were enumerated from six streams (three oiled and three unoled) and approximately 240,000 were tagged in both 1990 and 1991. Tags were applied to wild fry at ratios that ranged from one in three, to one in fifteen depending upon the total outmigration at each location.

Tag recovery rates vary by district, week, and processor (Peltz and Geiger, 1988). Tagging goals established in 1988, 1989, and 1990 were designed to ensure tags in subsequent adult returns could be recovered (F/S 3) in sufficient numbers to estimate the contribution of each release group to each district, week, and

processor stratum in the commercial fishery. This degree of precision was required to estimate differences in production for oiled and unoled groups of wild salmon (synthesis of F/S 1,2,3, and 28) and between wild and hatchery salmon. Tagging rates were held as constant as possible between and within hatcheries. An overall tagging rate of approximately 1 tag per 600 fish was chosen after analysis of the performance of previous tagging studies (Peltz and Miller, 1988; Peltz and Geiger, 1988; Geiger and Sharr, 1989).

Hatchery pink salmon fry to be tagged were randomly selected as they emerged from incubators. The fry were then anesthetized in a 1 ppm solution of Tricane Methanesulfonate (MS-222) prior to removal of adipose fins and application of tags. Tags were applied with a Northwest Marine Technology tag injector (model MKIV). A random sample of 20 clipped fish was graded for clip quality during each tagging shift. Clipped fish were tagged and passed through a quality control device (QCD) to test for tag retention. Rejected fish were retested at least three times and if repeatedly rejected, they were destroyed to minimize the number of untagged but clipped fish in the release. Fry which retained tags were held overnight to determine short-term tagging mortality. An overnight tag retention rate was estimated by randomly selecting 200 fish and testing them with the QCD before release into saltwater rearing pens. Tag placement was checked periodically but not quantified. At Prince William Sound Aquaculture

Corporation (PWSAC) hatcheries, unmarked fry entering the large saltwater rearing pens were counted with electronic fry counters. At the Solomon Gulch hatchery, the numbers of unmarked fry entering saltwater net pens were estimated from egg counts, with appropriate adjustments for egg mortality. At all facilities, pink fry mortalities were visually estimated immediately prior to release. These estimates were applied equally to tagged and untagged fish to obtain final release estimates. Fry releases were at night to reduce predation and were timed to coincide with peak plankton abundances near the hatcheries.

Methods of handling tagged fry prior to release differed slightly between PWSAC and Valdez Fishery Development Association facilities and also between years. In 1989, fry tagged at the Solomon Gulch hatchery were held for three days inside saltwater pens within the larger pens holding their unmarked cohorts. This allowed a short-term saltwater mortality estimate to be made which was then used to discount the total number of tagged fry released. At PWSAC facilities, tagged fry were released directly into saltwater pens containing unmarked fry. In 1990, tagged fry from PWSAC hatcheries were treated identically to those tagged in 1989. At the Solomon Gulch facility, only 500 marked fry were assessed for short-term saltwater mortality while the remainder were introduced directly into pens containing unmarked fish. In 1991, all tagged fry from all hatcheries were introduced into saltwater pens within the larger pens holding their unmarked cohorts allowing estimation of short-term saltwater mortalities for all facilities. One difference between facilities in 1991 was that

fry tagged at Solomon Gulch were held in freshwater incubators until all tagging within a single tagcode was completed, whereas fry tagged at PWSAC hatcheries were released into saltwater pens immediately after their 24-hour waiting period. Tagged fry at PWSAC facilities were thus introduced to saltwater in a staggered fashion. By deducting both the short-term tagging and saltwater rearing mortalities from the number of fry initially tagged, and accounting for overnight tag loss, the number of fry released with tags of each tagcode was calculated.

In Prince William Sound, intertidal spawning areas used by pink salmon contribute a majority of the wild pink salmon production (Helle et al., 1964). In 1990 and 1991, intertidal fry weirs were used to capture wild pink salmon fry emigrating from three oiled and three unoled streams in western Prince William Sound. Prior to the initiation of this oil spill study, attempts to tag wild pink fry with coded-wire tags had been limited to experimental work at Auke Creek in Juneau, Alaska (Johnson and Longwil, 1988). To capture the fry, winged fyke nets measuring 3.0 meters in height with 3.12 mm mesh openings (1/8 inch) were placed in the intertidal zones at the 1.8 m (6 foot) tide zone of each stream. The net was designed to fish a location subject to 3 meter tidal fluctuations and to capture fry as small as 27 mm. The fyke nets were attached to a floating live box designed to rise and fall with the changing tide. Captured fry were tallied using thumb counters or during the tagging process. Untagged fry were individually counted by placing a net-full of fry into a bucket and then slowly decanting the bucket over the stream and counting fry as they left the bucket. This method eliminated questionable measurement errors asso-

ciated with using average weights.

Half-length coded-wire tags were applied to a random sample of fry daily. Tagging was done at a rate coinciding with the daily out-migrations through each weir. Fry were captured in the intertidal zones of streams in constantly changing tide heights. They were then anesthetized with MS-222, adipose fin clipped and tagged in fresh water only, without any apparent negative effects. Working adjacent to the intertidal zone precluded the use of a submersible pump to provide water to the tagging site. Changing tides would put the source of water more than 70 meters away at low tide. Daily tidal fluctuations would result in fry being anesthetized and then recovering in water with constantly changing salinities.

To minimize osmoregulatory distress during the already stressful process of fin clipping and tag injection, a recirculating fresh water system was established. This closed system consisted of plastic trash barrels, fish totes and buckets with a capacity of approximately 60 gallons of water. The continuous movement of water through this system produced sufficient aeration. Water temperature was kept constant by placing snow and ice into plastic bags which were floated in the water.

In 1990, at the end of a tagging shift after all fry had visibly recovered from the anesthetic and the effects of tagging, all but approximately 200 fry were transferred to floating .5m x .5m x .5m net pens anchored in salt water. These tagged fry were held for 24 hours to obtain a count of short-term tagging mortality and then released. The 200 or so fry held in fresh water for 24 hours were tested for tag retention and also for short-term mortality prior to their being released.

Twenty-four hour survival rates of tagged fish at all sites were 99% for both years. Therefore, in 1991 tagged fry were held in freshwater to determine short-term tag retention and overnight mortalities and then released directly into saltwater along with their untagged cohorts. At Herring Creek in 1991, a second weir was installed above the mean high tide line to separately enumerate and tag the freshwater and intertidal components of the pink salmon outmigration. Tagging and enumeration methodology was the same as used at the intertidal weir sites.

Fry outmigrations began each year in mid-April and extended through early June in 1990 and through mid-July in 1991. Overall, outmigrating fry averaged .22 grams in weight and 31 millimeters in length. Beyond April, almost all captured fry had completely absorbed their yolk sac and were buttoned up. The removal of these newly emergent fry from the intertidal area, their transfer into fresh water for tagging and then back into salt water net pens for 24 hours did not result in detectable harm to the fry. For all six study streams, 24-hour survival rates of tagged fry were 99%. Smaller groups of tagged and untagged fry held for up to 72 hours exhibited similarly high survival rates. Twenty-four hour tag retention was also consistently above 90% at each site. These 24-hour survival and retention percentages compare very favorably with groups of pink salmon tagged at hatcheries.

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Impacts of the Exxon Valdez Oil Spill on the Migration, Growth, and Survival of Juvenile Pink Salmon in Prince William Sound

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This study focused on the effects of the Exxon Valdez oil spill on the growth, migration, and survival of juvenile pink salmon during the first two months of their marine residence in Prince William Sound. The number of adult pink salmon that return each year appears to be strongly affected by mortality during the early marine period (Parker, 1968; Ricker, 1976; Bax, 1983). During this time, slow growing individuals sustain a higher mortality, because they are vulnerable to predators for a longer time (Parker, 1971; Healey, 1982).

Oil contamination can reduce the growth rate of juvenile pink salmon in seawater (Moles and Rice, 1983). In the present study, the growth, migration, and survival of juvenile pink salmon was estimated from recovery of coded-wire tagged (CWT) fish released from hatcheries in Prince William Sound. In 1989, 1990, and 1991, approximately 500 million juvenile pink salmon were released each year from Prince William Sound hatcheries, and each year approximately one million of these fish were marked with a CWT inserted in their snout. Juvenile pink salmon were recovered with beach and purse seines in oiled and non-oiled nearshore habitats during the first two months after the fish were released from hatcheries.

The growth rates of juvenile pink salmon in Prince William Sound appeared to be affected by oil contamination from the Exxon Valdez. Growth rates of juveniles released from the Armin F. Koernig (AFK) Hatchery in 1989 were

significantly lower ($P < .003$) in the moderately oiled area near the hatchery than in the lightly oiled area along the southern coast of Knight Island. Growth rates of juvenile pink salmon released from the AFK Hatchery in these same areas were not significantly different ($P = 0.39$) in 1990. In 1991, growth rates were again significantly lower ($P < .001$) in the previously oiled area, but the magnitude of the difference was nearly one-half that observed in 1989. Growth rates of juveniles released from the Wally H. Noerenberg (WHN) Hatchery in 1989 were lower in the moderately oiled area along Knight Island Passage than in the non-oiled area near WHN Hatchery, but the difference was marginally significant ($P = 0.12$). Growth rates of juveniles released from the WHN Hatchery were not significantly different between these two areas in 1990 ($P = 0.30$) and 1991 ($P = 0.44$).

The lower growth rates of juveniles captured in the moderately oiled area near AFK Hatchery were likely caused in part by lower ocean temperatures in this area compared with the lightly oiled area along the southern coast of Knight Island. Results from regression analysis and analysis of covariance indicated that water temperatures were significantly different ($P = 0.041$) between these two areas. The mean difference in water temperature was 1.5°C . Results from laboratory experiments indicate that this difference in water temperature may cause a difference in growth rate of 0.32% body weight per day when juvenile pink

salmon are fed at maximum ration (Kephshire, 1976). The estimated difference in juvenile growth rates between these two areas of Prince William Sound was 1.62% body weight per day. Water temperatures were not significantly different between the moderately and non-oiled areas where juveniles from the WHN Hatchery were captured.

Stomach contents analysis was used to determine whether prey composition or apparent feeding rate was different between oiled and non-oiled areas. Samples of untagged juvenile pink salmon ($n=10$) were preserved at 64 oiled and non-oiled sites where CWT fish were captured in 1989. Prey organisms in each stomach were enumerated in the following categories: large calanoid copepods (>2.5 mm), small calanoid copepods (<2.5 mm), harpacticoid copepods, and "other prey." This approach was taken because the feeding rate of juvenile pink salmon is reduced when only small prey organisms are available (Parsons and LeBrasseur, 1973). Prey biomass in each category was estimated by the product of prey frequency and average prey wet weight. Total stomach contents weight was estimated by the sum of prey biomass in all categories. Wilcoxon rank-sums were used to test for differences in total stomach contents weight and prey biomass in each category between oiled and non-oiled areas.

Preliminary analyses indicated that biomass was significantly different ($P<.05$) in the large calanoid copepod, harpacticoid copepod, and 'other prey' categories between May and June. Separate Wilcoxon rank-sums were calculated for May and June to test for differences between oiled and non-oiled areas in these three prey categories. Total stomach contents weight and prey biomass in

all four prey categories were not significantly different between oiled and non-oiled areas.

The occurrence of P4501A-dependent monooxygenases was used to estimate the level of oil exposure of juvenile pink salmon in areas where CWT fish were captured (Stegeman et al., 1991). Samples were embedded in paraffin and thin sectioned. A monoclonal antibody that binds to P4501A-dependent monooxygenases was applied to each thin section. Monoclonal antibodies were detected by staining, and the occurrence of staining in various tissues was recorded. Separate chi-square analyses were conducted to test for differences in the frequency of occurrence of P4501A staining in gill, heart, and gastrointestinal tract tissues between sampling areas.

The frequency of occurrence of P4501A staining in gill tissues closely coincided with the degree of oil contamination observed in the sampling area. The frequency of occurrence of staining in gill tissues was significantly greater ($P=.003$) in the moderately oiled area near AFK Hatchery than in the lightly oiled area along the southern coast of Knight Island. The frequency of occurrence of staining in gill tissues was also significantly greater ($P<.001$) in the moderately oiled area along Knight Island Passage than in the non-oiled area near the WHN Hatchery. The frequency of occurrence of staining in heart and gastrointestinal tract tissues was not significantly different between oiled and non-oiled areas.

The migration of CWT juvenile pink salmon released from the AFK Hatchery may have been affected by oil contamination near the hatchery in 1989. One hundred and thirteen CWT juvenile pink salmon from AFK Hatchery were recov-

ered along the southern coast of Knight Island in 1989. Only 14 and 43 CWT juvenile pink salmon from AFK Hatchery were recovered in this area in 1990 and 1991, respectively. Visual observations of juvenile salmon abundance also indicated that much higher numbers of fish were present along the southern coast of Knight Island in 1989 than in 1990 and 1991. Laboratory experiments have demonstrated that juvenile pink salmon can detect and will avoid sublethal concentrations of oil (Rice, 1973).

Regression analysis was used to determine whether the growth rates of juvenile pink salmon in Prince William Sound were related to survival to the adult stage. Recoveries of adult CWT pink salmon in commercial fisheries and at hatcheries were used to estimate survival to the adult stage for each tag lot released from Prince William Sound hatcheries in 1989, 1990, and 1991 (Fish/Shellfish Study #3).

Results from regression analysis indicated that the mean growth rate of juvenile pink salmon by tag lot ($n=62$) was significantly related ($P<.001$) to survival to the adult stage. Analysis of covariance was used to test for homogeneity of slopes between years, hatcheries, and hatchery-treatment groups. The slope of the relationship between growth and survival was significantly different ($P<.001$) between years and marginally significantly different between hatcheries ($P=0.068$).

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The Impact of the Exxon Valdez Oil Spill on Juvenile Pink and Chum Salmon and their Prey in Nearshore Marine Habitats

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The objectives of this study were to determine the impact of the oil spill on juvenile pink and chum salmon during their initial period of marine residency in nearshore habitats. Field studies in 1989 and 1990 compared (1) exposure to and contamination by hydrocarbons; (2) distribution, abundance, size and nominal growth rates; (3) feeding habits; and (4) prey abundance for these fish between pairs of oiled and non-oiled locations in western Prince William Sound.

The dichotomous classification of our general sampling areas as "oiled" and "non-oiled" was substantiated by measurements of hydrocarbons in both mussel tissues and surface sediments collected in 1989. The degree of contamination in the oiled sites we sampled had greatly diminished in 1990.

Juvenile pink and chum salmon were contaminated by exposure to Exxon Valdez crude oil in 1989. There were detectable levels of hydrocarbons in tissues of juvenile pink salmon collected in the nearshore environment of oiled areas in 1989. In order to test that hydrocarbons detected in samples were not due to external contamination, flesh samples and viscera were processed separately from some samples of fish from oiled locations; both types of tissues were contaminated by hydrocarbons, with higher concentrations in the viscera. Exposure of both pink and chum salmon fry to physiologically significant levels of oil in 1989 was also indicated by induction of a mixed-function oxidase (MFO) enzyme,

cytochrome P4501A, in fry from oiled areas.

Ingestion, either of whole oil or oil-contaminated prey, was the principle route of contamination. Ingestion of oil was indicated by the distribution of the hydrocarbon in fry, the cell types induced for P4501A activity, and the presence of oil in the stomachs of 1% of the pink salmon and 3% of the chum salmon collected at oiled sites in 1989.

There was no evidence of continued contamination in 1990. Hydrocarbons were not found in the tissues of pink salmon fry from oiled or non-oiled areas, and P4501A induction was not observed in juvenile salmon the year following the spill.

Juvenile pink and chum salmon were more abundant in the non-oiled area in both 1989 and 1990. Because the pattern of abundance did not change as exposure levels diminished, we conclude that the differences observed in abundance were more likely due to geographic differences in the production and migration patterns of salmon fry rather than a response to exposure to oil.

Juvenile pink salmon moved rapidly from sheltered bays to more exposed, steep gradient beaches in migration corridors, where they fed predominately on zooplankton. This rapid movement is considered to be an adaptive feeding strategy in response to the distribution of zooplankton in nearshore habitats in Prince William Sound. The observation of this behavior over a wide geographic

range reinforces the conclusion drawn in the University of Alaska Fairbanks component of F/S-4, that the presence of oil-deflection booms in Port San Juan in 1989 disrupted the normal migration behavior of fish released from the Armin F. Koernig Hatchery (Cooney, 1990).

The diet composition and feeding efficiency of pink and chum salmon juveniles were not affected by the oil spill. Stomach fullness and numbers and biomass of prey consumed did not differ significantly between oiled and non-oiled areas. In 1989 consumption of epibenthic (sea floor) prey was greater in non-oiled areas and consumption of zooplankton prey was greater in oiled areas; the reverse pattern was observed in 1990. We attribute this switch in diet composition to differences in the timing and abundance of the spring zooplankton bloom.

Juvenile chum salmon in oiled areas may have been more susceptible to hydrocarbon exposure than pink salmon because of their distribution in nearshore habitats. Juvenile chum salmon utilized low gradient shorelines to a greater extent, and thus were more likely to forage over contaminated sediments. This habitat preference is also reflected in the higher utilization of epibenthic prey by chum salmon relative to pink salmon. The P4501A induction observed for chum salmon in oiled areas in 1989 was generally stronger than in pink salmon. However, juvenile chum salmon were generally rare in the oiled locations sampled.

Juvenile pink salmon grew significantly slower and were significantly smaller in oiled than in non-oiled locations in 1989. This analysis of unmarked fish corroborates the significant reduction in growth of tagged pink salmon recaptured in oiled areas reported in the Alaska Department of Fish and Game

component of F/S-4 (Willette, 1991). In 1990, there was no difference in size or growth of pink salmon fry between oiled and non-oiled areas. Condition factor tended to be higher for fry in oiled locations in both 1989 and 1990.

Juvenile chum salmon were significantly larger in the oiled locations in both 1989 and 1990. There was no evidence of a reduction in condition factor in the oiled area. Because chum salmon were rarely captured in oiled habitats, there were insufficient data to compare apparent growth rates for this species between oiled and non-oiled areas.

We found no evidence of a reduction in available prey organisms of juvenile salmon due to oil contamination. No significant differences were detected in the biomass of pelagic zooplankton between oiled and non-oiled areas in either 1989 or 1990.

Oil contamination did not reduce, and may have enhanced, the abundance of epibenthic prey species. Epibenthic prey biomass, which was composed primarily of harpacticoid copepods (small crustaceans), was higher in oiled locations than in non-oiled locations in 1989. This trend could have been due to geographic variability, reduced cropping associated with lower abundance of juvenile pink salmon, or direct enhancement by oil contamination. Enhancement by oil was indicated by the higher abundance of epibenthic harpacticoid copepods on heavily oiled than lightly oiled beaches in 1990. The preliminary analysis of the colonization of artificially oiled sediments imported to Prince William Sound in 1990 also showed a trend of higher abundance of harpacticoid copepods and other meiofauna (small animals) in oiled sediments relative to control sediments.

We conclude that the reduction in

growth observed for juvenile pink salmon in 1989 was caused by oil contamination. Temperature, prey availability, and feeding efficiency were as high or higher in oiled locations as in non-oiled locations in 1989, and therefore do not explain the observed reduction in growth. Condition of juvenile pink salmon was generally higher in the oiled habitats. These fry were feeding and growing in the oiled habitats, but losing growth efficiency due to the metabolic cost of depurating the hydrocarbon burden. The reduction in growth of juvenile pink salmon due to exposure to hydrocarbons in the marine environment was limited to the first year after the spill. We found no evidence of measurable contamination or physiological effects (e. g., apparent growth reductions, P4501A induction, or hydrocarbon body-burden) in 1990.

It is likely that there was some incremental reduction in the potential survival of pink salmon juveniles contaminated by oil in 1989. Within a year-class, slower-growing groups of pink salmon fry have lower marine survival than their faster-growing cohorts (Mortensen et al., 1991). Fry migration studies have shown that most of the hatchery and wild pink salmon produced in Prince William Sound move seaward through the oiled areas (Willette, 1991). Thus large numbers of juvenile salmon, including populations originating from outside the actual spill area, were exposed to hydrocarbon contamination in the marine environment during their migration to the Gulf of Alaska.

Such large scale exposures, linked with growth reductions, may have resulted in a large scale reduction in the

overall recruitment of pink salmon to Prince William Sound in 1990.

The effects observed for pink salmon in 1989 could have also occurred in other species. Chum salmon juveniles captured along oiled beaches showed significant cytochrome P4501A induction. A wide variety of other fishes utilize the nearshore waters of Prince William Sound and the adjacent Gulf of Alaska, and many pelagic schooling fishes and larval fishes utilize zooplankton as their principal prey (Rogers et al., 1986). If ingestion of either whole oil or contaminated prey was the route of exposure for juvenile pink salmon, then a large number of other fishes with similar feeding habits may also have been contaminated.

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Effect of Oil-contaminated Food on the Growth of Juvenile Pink Salmon, *Oncorhynchus gorbuscha*.

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To test the hypothesis that ingestion of oiled food affects the growth and survival of juvenile pink salmon (*Oncorhynchus gorbuscha*) we fed them food contaminated with crude oil. This experiment resulted from the observation that juvenile pink salmon collected from locations impacted by the Exxon Valdez oil spill in Prince William Sound, 1989, were contaminated with oil. Ingestion of oil was the probable route of contamination for these fish.

Our objectives were to (1) measure mortality; (2) investigate the effect of ingested oil on growth (length and weight); (3) determine if otolith growth (whole axis and radial distance) and increment periodicity could be used as indicators of metabolic stress induced by exposure to ingested oil; (4) observe effects of oiled food on feeding, measured by percent stomach weight and fecal production; (5) measure hydrocarbon uptake in tissues; (6) measure cytochrome P4501A enzyme induction; (7) determine histological response; and (8) determine whether nucleic acid measurements (RNA and DNA) could be used to evaluate the effects of crude oil on juvenile salmon after an oil spill. In this paper we present an overview of the experiment. Three other papers dealing with specific topics are in press or in preparation: a nucleic acid paper (Wang et al., in press), an otolith paper (Mortensen and Carls, in prep), and a paper dealing with hy-

drocarbon uptake, cytochrome P4501A induction and histological response (Carls et al., in prep).

Juvenile pink salmon (*Oncorhynchus gorbuscha*) were obtained from the Auke Creek Hatchery, Juneau, Alaska, on April 16, 1991 and reared in seawater for 3 weeks. Four days prior to treatment, fish were randomly allocated into five treatment groups, with three replicate tanks per group. Each 30 x 41 x 53 cm tank received 1.4 l/min of seawater filtered through a 25 μ m filter to ensure that potential prey was not introduced into the tanks. The initial number of fish per tank averaged 1076; initial fish size was 34.3 ± 0.3 mm and 242 ± 8 mg. Mean water temperatures were maintained at $7.8 \pm 0.02^\circ\text{C}$. At weekly intervals, 100 fish from each tank were sampled at random for analysis. After six weeks of treatment (May 13 - June 26), all groups were offered untreated food for two additional weeks to investigate possible recovery.

The fish were fed 0.6 - 0.8 mm diameter Biodiet¹ pellets in excess (10% of the estimated biomass/tank/day). Concentrations of Alaska North Slope crude oil in the food were 0.00 and 0.02 ± 0.05 mg/g for controls and treated controls, and 0.37 ± 0.04 , 2.78 ± 0.25 , and 34.83 ± 5.4 mg/g for low, mid, and high oil treatments, respectively. The purpose of the treated control was to determine if dichloromethane, used as a carrier sol-

vent to facilitate the mixing of crude oil in the food pellets, affected the nutritional quality of the Biodiet feed. Treated control concentrations did not differ significantly from control concentrations.

Hydrocarbons accumulated in the tissues of pink salmon exposed to oiled food. Accumulated concentrations were directly correlated to concentrations in the food, both in carcass tissues and in viscera tissues that were devoid of food and fecal matter. Concentrations of hydrocarbons in viscera exceeded those observed in carcasses.

Consumption of oiled food caused mortality in the highest treatment; differences became significant after two weeks of exposure, and increased rapidly until fish began feeding on clean food. This mortality was probably due to direct oil effects and starvation. Fish in the high treatment continued to exhibit strike behavior and consume food throughout the six week treatment period, but feeding rates, measured by stomach content and fecal production, were inhibited within the first week of exposure. When offered clean food, mortality ceased abruptly, growth and feeding rates increased, and swimming behavior became more normal. Mortality in all other treatments, including controls, remained low (about 4%) throughout the six week treatment period.

Somatic and otolith growth were reduced as a consequence of consuming oil-contaminated food. Somatic growth was significantly ($P \leq 0.05$) reduced in the mid and high treatments within one week and in all oil treatments after two weeks. Otolith growth was significantly depressed by all oil treatments within the first week of the experiment. The number of increment rings deposited was also reduced by contaminated food.

Energy available for growth could have been reduced by metabolic demands, reduced digestive assimilation efficiency, and, at high oil concentrations, reductions in feeding. Preliminary analysis of cytochrome P4501A showed these enzymes were induced, indicating expenditure of metabolic energy occurred.

Although feeding rates actually increased in the low oil treatment, and generally did not decline in the mid oil treatment, significant depression of growth was observed in both groups. This suggests that changes in growth were not simply due to starvation, but rather were due to metabolic demand or reduced assimilation efficiency. Necrosis of the gastrointestinal tract was observed in the single low treatment sample analyzed to date, so a physiological mechanism for reduced assimilation efficiency may exist.

Ingestion of crude oil-contaminated food had clear dose and time dependent effects on the nucleic acid content (amount of nucleic acid in μg per mm fork length), and concentration (μg nucleic acid per mg dry weight) on juvenile pink salmon. Nucleic acid content increased in controls but increased more slowly or not at all in the mid and high treatments, leading to significant separation. DNA concentrations in the high oil treatment increased significantly after one week, and remained elevated thereafter. In the high oil treatment, RNA concentration decreased significantly during the first week, but increased to control levels during the subsequent five weeks. Although somatic growth occurred in all treatment groups, it was achieved at the expense of decreased cellular content among fish fed high dose food. After contaminated food was replaced with clean food, there was a sig-

nificant increase in RNA concentration and a decrease in DNA concentration, although DNA concentration in the high oil treatment remained substantially greater than in controls. RNA content ($\mu\text{g}/\text{mm}$) and RNA/DNA ratio both correlated significantly with growth ($r^2 = 0.933$ and $r^2 = 0.478$, respectively) and thus could serve as indicators of growth in fish collected from the wild.

In summary, hydrocarbons accumulated in juvenile pink salmon that consumed oiled food, somatic and otolith growth was reduced, biochemical changes occurred (in nucleic acid concentrations and content, and induction of P4501A enzymes), histological problems developed (necrosis of the gastrointestinal tract), and, at high concentrations, mortality became significant. Feeding rates increased at low oil concentrations, but were depressed at high concentrations. Energy available for growth could have been decreased by metabolic demands, reduced digestive assimilation efficiency, and, at high oil concentrations, starvation. Results of this experiment support the hypothesis that ingestion of crude oil was the prob-

able route of contamination in juvenile pink salmon captured in Prince William Sound following the *Exxon Valdez* oil spill, and that this contamination caused the growth reductions observed in fish captured in oiled habitats.

¹Reference to trade names does not imply endorsement by the National Marine Fisheries Service, NOAA

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Coded Wire Tag Recovery Results for Prince William Sound Pink Salmon, 1989-1991

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This study is one of an integrated group of Natural Resource Damage Assessment Fish/Shellfish Studies (NRDA F/S Studies) conducted to quantify the damage inflicted upon the pink salmon resource of Prince William Sound by the *Exxon Valdez* oil spill. The studies associated with pink salmon, namely the egg-fry mortality study (NRDA F/S Study 2), the early-marine survival study (NRDA F/S Study 4), and the adult survival studies (NRDA F/S Study 3) were designed to document possible effects of the spill at various stages in the life cycle.

The integration of results from studies 1 through 4 will provide stock-specific estimates of post-spill population size for oil-impacted as well as unimpacted stocks and will also identify possible sources of population level damage inflicted by the spill at intermediate life stages. The Run Reconstruction study (NRDA F/S Study 28) will attempt to reconstruct historic returns by stock for comparison with post spill returns as a means of assessing overall population damage. The coded wire tag program provided the data pertaining to the commercial and cost-recovery catches of hatchery and wild fish, which when combined with escapement data (NRDA F/S Study 1), provided the post-spill population estimates of wild returns. The program also provided fish of known origin (i.e. oiled versus unoiled area) for the early marine survival and adult-survival studies. By allowing estimates of contributions to the fishery to be made on an area-time basis, it provided essential data

for the Run Reconstruction program.

Coded wire tags were applied to pink salmon fry at four Prince William Sound hatcheries in 1988, 1989, and 1990 and to wild fry at three oiled and three unoiled streams in 1990 and 1991 (Loomis, Cathead, Herring [oiled streams], Totemoff, O'Brien and Hayden [unoiled streams]). Tagging rates were such that recoveries of marks in subsequent adult returns allowed estimation of the contribution of each release group to each district, week, and processor stratum in the commercial fishery with an acceptable degree of precision. An acceptable degree of precision is one which allows detection of meaningful oil-induced effects and which yields contribution estimates which are useful to fishery managers. Tagging rates at the hatcheries were held at a rate of 1 tag per 600 fish. Wild stock tagging rates ranged from 1 tag per 5 fish to 1 tag per 20 fish.

Tags were recovered from adults caught in the common property and cost-recovery fisheries. Stratified random samples were taken from the catches made during 1989 through 1991, the strata being formed from district, week and processor combinations. The tendency of processors to concentrate their buying in certain areas of a district (Peltz and Geiger, 1990) necessitated the latter stratification. Technicians sampled adult salmon as they were pumped from tenders onto conveyor belts at various processors located in Cordova, Valdez, Whittier, Anchorage, Kenai, Seward and Kodiak during each district/period open-

ing. Time and location of catches were determined from processor tender logs and fish sales receipts (fish tickets). Technicians used visual and tactile methods to scan the fish for missing adipose fins, indicating that a coded wire tag probably resided in the snout. Harvest data and number of fish scanned were recorded for each tender load of fish and heads of adipose-clipped fish were sent to the Juneau Tag Lab for tag removal and decoding.

Tags were also recovered from adults in the brood stock at four Prince William Sound pink salmon hatcheries during egg takes. After the salmon were manually spawned, technicians used visual and tactile methods to scan approximately 95% of the fish. Brood stock scanning is an important part of estimating hatchery contributions. Due to differential mortality between tagged and untagged fish as well as differential tag loss between release groups, the tag expansion factor at release does not accurately reflect the tag expansion factor in the adult population. Theoretically, the brood stock is 100% hatchery fish. Therefore, tag recovery rates found in the brood stock were used to adjust the initial tag expansions for each hatchery.

Escapement tag recoveries fulfilled the same function as the brood stock recoveries. The tag rates found from scanning carcasses on stream surveys were used to adjust the initial release tag expansions. Carcasses were scanned for coded wire tags at the six tagged wild stock streams. Only carcasses with a visible adipose region were counted. Total number of carcasses and total number of adipose-clipped fish were recorded on a daily basis for each stream surveyed.

The number of tag recoveries along

with the catch information and total-release data were used in generating contribution and survival estimates. The total hatchery contribution to each harvest type was calculated as the sum of the estimated contributions of all release groups over all week, district and processor strata. A variance approximation for this quantity, which ignores covariance between release groups, was derived for sampled strata by Geiger (1990), and used to calculate appropriate confidence intervals. The average tag recovery rate for all processors in a week and district was used to estimate hatchery contribution in catches delivered to processors not sampled in that district and week. Variances associated with unsampled strata were not calculated. They are believed to be small.

Hatcheries contributed 83% of the pink salmon catch (18 million fish) in 1989, 70% (32 million fish) in 1990, and 84% (31 million fish) in 1991. The total contribution by hatchery for the three years was 15% (16 million fish) for Armin F. Koernig, 29% (30 million fish) for Wally H. Noerenberg, 16% (17 million fish) for Cannery Creek, and 17% (18 million fish) for Solomon Gulch. Wild pink salmon contribution to the fisheries was estimated by subtracting the hatchery contribution estimate from the total catch. When contribution data were combined with escapement data, post-spill returns at the population level were made available. These are key components in a major part of the damage assessment study, as are district-time contribution data in the Run Reconstruction project.

Recovery results were also used to estimate survival rates for hatchery fish and for wild fish originating in oiled and unoled streams. Survival rates for each tag release group were calculated by

dividing total estimated returns for that group by the total release of that group. Appropriate confidence intervals for the survival rates were generated.

Overall hatchery survival rates were 3.4%, 6.2% and 5.2% for fish recovered in 1989, 1990, and 1991, respectively. The ranking of the survival rates associated with the A.F. Koernig hatchery (an oiled facility) fell from second for the 1989 recovery year (fish released prior to spill) to fourth for the 1990 and 1991 recovery years (fish released during or after spill). Survival rates of fish from Cannery Creek (un-oiled facility) also fell from the 1989 recovery year to the 1990 recovery year, and moreover, fell from first place to joint last. Little could be concluded from these survival data.

The mean survival rate for fish migrating from oiled streams during peak migration was 2.8%, and the survival rate for those migrating after the peak was 0.77%. The survival rates for fish leaving un-oiled streams over the same periods were 2.3% and 1.3%, respectively. A significant interaction between the oiling and timing factors was found ($p=0.012$). It appears that the influence of migration timing is different for oiled and un-oiled streams. The influence of

timing on survival rates for oiled streams is between 0.53 and 1.47 percentage points greater than that for un-oiled streams (95% confidence).

These survival data constitute an essential component in the overall damage assessment program. In the management context, contribution estimates made in the various districts and weeks were, where possible, presented to managers attempting to provide migration corridors for the less abundant wild salmon. In this way, the coded wire tag program continued to function in its original capacity, as a management tool.

When combined with results from other damage assessment studies (NRDA F/S Study 1, 2, 3, 28), further and more penetrating determinations of possible oil-induced damages to the pink salmon resource will be made.

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Pink Salmon Spawning Escapement Estimation in Prince William Sound

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Each year approximately 400 million wild pink salmon *Oncorhynchus gorbuscha* fry migrate from hundreds of streams bordering Prince William Sound and each summer an average of 8 million adult wild pink salmon return to those same streams to spawn and die. Both juvenile and adult salmon play a major role in the Prince William Sound ecosystem. Salmon are an important prey species for a variety of other fish and birds as well as marine and terrestrial mammals. As prey items and as decomposing carcasses, adult salmon provide a vital pathway of nutrients from the marine ecosystem to the nearshore and terrestrial ecosystem.

In recent times, adult returns of wild pink salmon have contributed 6 to 7 million fish annually to the region's commercial fisheries. Accurate stream-specific estimates of wild pink salmon spawning escapement are essential to evaluate population damages from the *Exxon Valdez* oil spill. Accurate enumeration of spawning escapement also provides fisheries managers with information necessary to protect impacted stocks from excessive harvest in fisheries dominated by healthy wild and hatchery stocks.

This study is a part of an integrated group of Natural Resources Damage Assessment Fish/Shellfish Studies (NRDA F/S) conducted to quantify damage to pink salmon populations as a result of the *Exxon Valdez* oil spill. Some of these studies were designed to identify injury at specific life history stages while others were designed to quantify population level damages. Assessment of

damage to wild populations cannot be completed without accurate stock-specific estimates of total adult returns in both the pre-and post-spill era. Results from this study will be used in conjunction with catch contribution data from NRDA F/S Study 3 to estimate post-spill returns for oil-impacted stocks. The results will also be used by another NRDA study (F/S Study 28) to reconstruct pre-spill returns for damaged pink salmon populations. Finally, more accurate estimates of pink salmon escapements will be used by resource managers to refine harvest management strategies and direct fishing effort away from damaged stocks which are receiving inadequate escapement.

Historically, counts of spawning fish obtained from a systematic weekly aerial survey program and stream residence time (stream life) data from one large stream in eastern Prince William Sound have been used to estimate pink salmon escapement for 209 streams in Prince William Sound (described by Johnson and Barrett, 1986). This method of estimating escapement did not account for observer bias, inter-stream variations in stream life, temporal variations in stream life, or missing counts from unsurveyed streams. The purpose of this study was to (1) estimate aerial observer bias through comparisons of aerial counts to weir and foot survey counts on a variety of stream types, (2) estimate temporal and spatial variations in stream life for a subset of the 209 streams surveyed aerially, and (3) use observer bias and stream life data to estimate current year and

historic escapements for the 209 streams in the aerial survey program.

Weirs were placed across one large stream in eastern Prince William Sound and three smaller streams in western Prince William Sound in 1990. The same four streams plus six additional small to moderate sized streams in western Prince William Sound were weired in 1991. All weirs were placed at the lower limit of intertidal spawning (approximately 1.8 m above mean low water). The total upstream migration to spawning areas was enumerated daily through each weir.

Foot surveys were conducted on each weired system in 1990 and 1991. Daily foot surveys were also conducted on 36 unweired streams representative of a variety of stream types. Foot survey crews reported daily counts of live and dead salmon by species for five zones in each stream; four intertidal zones between elevations of 0.0-1.8 m, 1.8-2.4 m, 2.4-3.0 m, and 3.0-3.7 m above mean low water; and the upstream zone above mean high water to the upstream limit of spawning. Dead salmon were marked to prevent duplicate counting in subsequent surveys.

Numbers of live fish in the stream on any given day were estimated from weir data and foot survey counts of dead fish. This estimate was compared to aerial survey counts to estimate aerial survey bias. At weired streams, the number of live fish present on day i was estimated as the cumulative count of fish through the weir on day i minus the cumulative count of dead fish above the weir on day i . For streams in the foot survey program, the number of live fish present on day i was simply the foot survey estimate of live fish on that day.

Mean stream life by week and the associated variance were estimated by

Peterson disk tagging studies. Tags were applied at weekly intervals to fish entering the mouth of each stream. Tags were uniquely colored to represent date of tagging, uniquely lettered to identify the stream where applied, and uniquely numbered for identification of individual fish. Weir and foot survey crews recorded the date of entry and death of tagged fish. Stream life was estimated for each tagged fish by the difference between the date of death and the date of entry into the stream.

Mean stream life was also estimated using two methods which did not rely on tag data. The first method estimated mean stream life by dividing the cumulative live fish days for a stream by the estimated total escapement where total escapement was estimated by either the cumulative weir count or the cumulative dead count for unweired streams. The second method estimated mean stream life as time elapsed between the mean date of abundance of newly arrived live fish (mean date of arrival) and the mean date of abundance of fresh carcasses (mean date of death).

Estimates of live fish in streams using aerial and ground survey methods were compared with data from weirs. The correlations between weir data and concurrent ground survey estimates in 1990 ($r^2 = 0.893$, $n = 31$) and 1991 ($r^2 = 0.949$, $n = 92$) were closer than a similar comparison between weir-derived and aerial survey derived estimates ($r^2 = 0.572$, $n = 36$ and $r^2 = 0.495$, $n = 92$). Both methods were biased; ground surveyors observed approximately one half of fish in the stream in 1990 and 1991 while aerial surveyors viewed approximately one-half of the true number of fish in 1990 and one-third of the true number in 1991. In general the bias increased as numbers of

salmon increased and was greater on long streams with significant upstream spawning. The larger bias in aerial counts in 1991 is consistent with the larger percentage of upstream spawners observed in the odd year cycle in Prince William Sound. Upstream channels are generally more convoluted and the forest canopy precludes good visibility.

Average times from date of tag application to date of tag recovery from dead fish were not significantly different between 1990 and 1991 (15.1 days versus 13.9 days). Milling of tagged fish at the stream mouth was not accounted for in 1990 and the mean time of 15.1 days from date of tagging to date of death is longer than the true stream life. When milling time at stream mouths was accounted for by using date of entry and date of death data in 1991 the average stream life estimated for all streams ranged from 3.2 to 19.6 days and averaged 9.8 days. Mean stream life estimates from other methods differed only slightly but were higher than those from tagging results.

Estimates using cumulative fish days through weirs averaged 10.5 days while estimates using the mean date of entry into weired stream and mean date of death averaged 11 days. If foot survey data were used instead of weir data to estimate live fish entry into streams, stream life estimates were approximately one day shorter than indicated by tagging results.

Pink salmon escapement through four weirs in 1990 was 72,244 fish. Escapements through the same weirs in 1991 totaled 159,748 fish. Escapements to these four streams in 1990 and 1991 were greatly underestimated (-41% and -75%) when the traditional method and stream life value were applied to aerial survey data. When the escapement estimates

from aerial data were adjusted for surveyor bias and recent stream life data, the negative bias of the aerial method was reduced to -28% in 1990 and to -17% in 1991. The greater bias observed in 1990 is due in part to using stream life estimates which are uncorrected for milling time at the stream mouths.

Surveyor bias and stream life estimates for the various stream types in this study can be applied to similar streams and used to improve escapement estimates for all 209 streams in the aerial survey program. Because they differ, surveyor bias and stream life must be estimated independently for the odd and even year cycles.

Analyses of 1990 and 1991 aerial data which incorporate counts corrected for surveyor bias and new stream life data are preliminary but indicate that traditional escapement estimates for many Prince William Sound streams may be far too low. The sources of bias in escapement estimates vary geographically. In eastern Prince William Sound long streams with significant upstream spawning are prevalent, updated stream life values are similar to the 17.5 day value used traditionally, hence escapement estimation bias is mostly a function of large observer bias.

Conversely, in western Prince William Sound streams are short and steep and intertidal spawning predominates. In these streams there is less observer bias but the 17.5 day stream life used traditionally is much too long and does much more to bias escapement estimates.

Given the major role which wild pink salmon play in the Prince William Sound ecosystem and the risks these populations now face from habitat destruction, over harvest in fisheries dominated by hatchery returns, and possible genetic

dilution from hatchery stock straying into natural systems, it seems imperative that we document any additional risk that these stocks may have inherited as a result of damage from the *Exxon Valdez* oil spill.

It is also imperative that we continue to provide these populations an added measure of protection until this potential new source of risk has been quantified or discounted. Accurate escapement estimation procedures are vital for damage assessment and enhanced management of Prince William Sound pink salmon

populations.

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A Life History Approach to Estimating the Damage to Prince William Sound Pink Salmon From the Exxon Valdez Oil Spill

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The precise effects of the Exxon Valdez oil spill will never be known. In the case of pink salmon within Prince William Sound, information was collected that can be used to develop a reasonable inference. Following the oil spill, studies were initiated to (1) develop improved estimates of salmon escapement for pre- and post-spill years, (2) measure egg survival, (3) measure fry survival, (4) observe the nearshore condition and distribution of fry in Prince William Sound, (5) measure hatchery and wild salmon survival in the saltwater environment, and (6) survey the pink salmon habitat affected by the oil spill.

To estimate the effects of the oil spill on Prince William Sound pink salmon, we followed the 1988 brood year of pink salmon, from their emergence in 1989, to their migration through the oil in the nearshore environment in the spring of 1989, to their return in 1990. We then continued with the odd-year line, following the 1989 brood year. These fish incubated in the oiled and unoled streams in Prince William Sound and returned in 1991. At each lifestage, we summarized the evidence of an oil effect on survival by means of probability distributions. These estimates were based on a life history approach, coupled with an analysis of the mechanisms of damage to the pink salmon resource.

A central issue that we will need to continue to refer to is that of *compensatory*

mortality (i.e. mortality that controls population size by increasing when density is high, and decreasing when density is low). The mechanisms of compensatory mortality could be such things as overtaxing the food supply or disease outbreaks triggered when the organisms become too crowded. If compensatory mortality effects are negligible at a particular lifestage and all subsequent lifestages, then estimating the effects of the oil spill are straightforward: The estimated proportional loss of production is simply the measured decrease in estimated survival at that lifestage. Alternatively, if compensatory mechanisms exist, the loss of production from the oil spill will be proportional to the decrease in survival, adjusted, in some fashion, for the magnitude of compensation.

Compensatory Mortality

Does there appear to be compensatory mortality in pink salmon at moderate escapement levels? If so, at which lifestages does compensatory mortality occur? A consensus is emerging that compensatory survival effects in pink salmon, if they exist, occur in the freshwater and early marine lifestages. For example, Alexandersdottir (1987) states

"Compensatory mortality appears to be a factor in the lifestage from spawning to outmigration. Heard (1978) reported on low year class survival of fry in Sashin Creek in

Southeastern Alaska in 1967, where he suggested compensatory mortality was due to an overabundance of spawners. Donnelly (1983) found that density-dependent factors were significant in the survival of pink salmon in Kodiak to outmigration."

Heard (1991) and Ignell (1988) have further information on the mechanisms and the magnitude of compensatory mortality in the freshwater stages.

The effect of density on pink salmon in the marine environment appears to be unrelated mortality. Density in the marine lifestages does affect growth and size, which affects fecundity (Heard 1991). In Prince William Sound, the hatcheries provide strong evidence that the number of fry emigrating from freshwater systems are linearly related to the number of returning adults. The relationship between fry released and adult recruitment in Prince William Sound hatcheries has been observed over the range of 0 to approximately 500 million fry, with no evidence of compensatory survival mechanisms in the marine lifestages. Although the estimates from the early years of this series were somewhat speculative, the estimates from years with large releases are based on coded wire tags. The slope of a smooth line through this data is approximately 5%, a fully reasonable average survival value based on other pink salmon populations (Heard, 1991).

Basic Life History Population Dynamics

The fisheries management apparatus in Prince William Sound has collected some of the most consistent, reliable, and well organized data relating to pink salmon population dynamics in the world. However, these data are actually of limited use in quantifying these popu-

lation dynamics in the Sound. The pink salmon fishery is managed to achieve a fixed escapement level; for that reason, the historical record contains very little information about pink salmon recruitment outside of a small escapement range. While recruitment is usually thought of as a function of stock size at a previous step (e.g. Ricker, 1975), the data contain large observational error in the stock size relative to the subsequent error in observing recruitment. This error-in-predictor problem will lead to potentially disastrous biases (Draper and Smith, 1981) if usual least-squares approaches are used to estimate stock-recruitment parameters based on the incorrect values of stock size.

Accordingly, we used Kalman filters and various robust nonparametric methods to estimate the relationship between potential egg deposition (PED) and subsequent observed recruitment.

Mortality Effects in the Freshwater Stage

In this lifestage, the mechanism by which the oil spill reduced survival is direct mortality. Up to 75% of the wild pink salmon which spawn in Prince William Sound use intertidal areas (Helle et al., 1964). These areas are highly susceptible to contamination from marine oil spills. Moles et al. (1987) and Rice et al. (1975) found that pink salmon eggs and pre-emergent fry were affected by exposure to crude oil and that the effect was most acute in intertidal environments.

Estimates of direct mortality and the proportion of eggs affected are taken from other studies reported at the symposium. We examined 1) the immediate effects of oil exposure on pre-emergent pink salmon numbers in the spring of 1989, 2) the effect of intertidal oil exposure on pink salmon egg mortality, and

3) the effect of intertidal oil exposure on pink salmon egg to pre-emergent fry survival. Thus we are provided a basis for estimating the reduced survival through direct mortality.

Mortality Effects in the Nearshore Stage

Mortality in nearshore areas can result from several mechanisms. Petroleum is toxic to pink salmon fry if ingested (Moles et al., 1987). However, various authors have shown that crude oil and petroleum components will reduce the growth rates of juvenile salmonids (Woodward et al. 1981; Moles et al. 1987; Vigner et al., 1992). In particular, Moles and Rice (1983) have described reduced weight and growth rate in pink salmon exposed to a 40 day dose of 33% of the 96-h LC_{50} . Aside from direct toxic effects of petroleum products, elevated mortality in the nearshore environment can be predicted from decreased growth rates that cause juveniles to remain at size stages with increased predation. In Heard's (1991) review of pink salmon life history, he discusses predation of pink salmon and concluded that mortality is highest in the early marine environment; Heard also cites others that concluded that mortality in juvenile pink salmon decreases with size. Parker (1962a, 1962b, 1964, 1965, 1968, 1971) has studied the relationship of pink salmon size to natural mortality, and his work forms the basis for estimated losses at this lifestage.

Survival Effects in the Oceanic Stage

Survival in the oceanic lifestage is assumed to be random, and unrelated to density in the freshwater and nearshore environment, and independent of oiling level. This assumption is in agreement with much of the literature (discussed in the section on the nearshore lifestage), and is in complete agreement with the

available information from Prince William Sound hatchery survival patterns.

The pink salmon in Prince William Sound represent different things to different people: They are an important input to the fishing industry, and therefore have an important economic role; they also play an important role in the terrestrial and aquatic ecology, and therefore have an important biological role. We have estimated the effects of the oil spill from a fisheries perspective, using biological models to describe the effects of the oil spill to the fishery resource.

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Overescapement Impacts of Kenai River Sockeye Salmon

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Studies conducted on the Kenai River system have provided new insight into the effects of the 1989 *Exxon Valdez* oil spill fisheries closures on production of sockeye salmon. Overescapements of adult sockeye into the Kenai River during 1987, 1988, and 1989 resulted in a declining trend in smolt production; smolt outmigration in 1991 was three million, followed by less than 500,000 in 1992.

Densities of juvenile sockeye in Skilak Lake, as indicated by a hydroacoustic survey conducted in May of 1992, were less than 10% of those observed during the fall of 1991. This is approximately the decline observed from the 1990 fall fry to 1991 smolt, during the previous years investigations.

Fall fry length and weight data from Skilak Lake, Kenai Lake, and Tustumena Lake have been collected over time and indicate a significant decrease in length and weight when compared to earlier samples from 1986.

All of the Kenai fry and smolt data from the studies over the past several years are internally consistent, and support the hypothesis that overescapements during 1987, 1988, and 1989 have adversely affected the sockeye smolt production from the Kenai River system.

The investigations of the cause of the decline have centered on the availability of zooplankton to juvenile sockeye. Biomass of zooplankton in Skilak Lake decreased during the past two years, but the decrease was moderate compared to the decline in fish production. Diel

vertical migration (DVM) of zooplankton observed in the spring and summer of 1992 in Skilak Lake was highly accentuated, particularly for egg-bearing copepods. These phenomena were not observed in Tustumena Lake, a nearby glacial system which has not experienced a major collapse in sockeye salmon production or overescapement. We investigated zooplankton in two additional glacial lakes within the Kenai River drainage (Grant and Ptarmigan lakes), because they did not have significant sockeye salmon populations. These lakes did not demonstrate the degree of DVM displayed in Kenai and Skilak lakes. More intensive investigations as to the density-dependent mechanism that has created catastrophic declines in sockeye smolt production from this system are currently being conducted.

If the hypothesized change in the plankton community proves to be the density-dependent mechanism responsible for the poor smolt production, restoration of the plankton community to a more productive state would be desirable. Other authors suggest that these changes may be induced behaviorally and subsequently may recover instantly once the stimulus is removed (Stirling, 1990). The most likely stimulus removal could occur naturally by predator reduction through lowering of sockeye spawners in the future. The availability of zooplankton would probably increase as food limitations would provide an adequate stimuli to promote increased time

of surface feeding. This would be expected to occur when the system converts from top-down control over the system's trophic state to bottom-up control as sockeye predation is reduced (Carpenter et al., 1985). Other investigators have experimentally established food reduction as a method of increasing the surface time or elimination of the DVM response (Johnsen and Jakobsen, 1987). Alternatively, this response behavior may be caused by natural selection of those individuals which have the propensity to migrate, probably at the expense of reproduction. Genetic differences have been demonstrated in *Daphnia* between individuals exhibiting DVM and those which do not (Weider, 1984). We know some genetic variation occurs in the *Diaptomus* taxa in the Kenai River system because of a pigmented racial variate which occurs in Ptarmigan Lake. In this and other lakes which are absent of planktivores, a red carotinoid pigment is present in virtually all individuals within

these subpopulations of this taxa. This pigment apparently is an adaptation to harmful near-surface sun radiation (Hairston, 1976). We would expect significantly longer recovery times if the sockeye production has been reduced because of genetic changes in the zooplankton community.

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Effects of the T/V Exxon Valdez Oil Spill on Murres: A Perspective From Observations at Breeding Colonies

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Murres (*Uria* spp.) were congregating near their breeding colonies in the western Gulf of Alaska when the T/V Exxon Valdez oil spill occurred in March 1989. Tens, perhaps hundreds, of thousands were killed (Piatt et al. 1990). In order to evaluate the impacts of the oil spill on murres, we surveyed murre breeding populations at five locations where there was direct evidence of oiling (Chiswell Is., Barren Is., The Triplets, Puale Bay, and Ugaiushak I.) and at two locations just outside the trajectory which were probably not affected by oil (Middleton I. and Chowiet I., Semidi Is.). Most sites were surveyed annually from 1989 to 1991. In addition to counts of birds, we gathered information at some of the sites about the timing of nesting events and productivity.

Historical data on murres at most colonies in the Gulf of Alaska were obtained during brief visits which necessitated single counts or crude estimates, and survey methods were seldom clearly documented. In contrast, we counted murres after the spill during hours and dates when variability in attendance at colonies is minimal (Byrd 1989, Hatch and Hatch 1989) and under observing conditions that minimized observer error. Furthermore, replicate counts were made at most sites to improve estimates. Annual counts of murres from 1989-1991 at colonies within the trajectory of the oil were 40% to 60% lower than pre-spill counts, whereas counts at colonies just outside the trajectory did not decline following the spill.

Despite uncertainties about the accuracy of historical counts, significant differences in numbers present before and after the spill indicated definite declines. Only the magnitude was equivocal. Since populations of murres at colonies just outside the trajectory of the oil did not decline following the spill and direct mortality within the trajectory was so pronounced, it seems likely that oil mortality caused the population declines observed at affected colonies.

Mortality is not the only possible cause of reduced counts at murre colonies. In cases where environmental perturbations are severe (e.g., El Niño Southern Oscillation), food webs can be so disrupted near colonies that many murres abandon cliffs during the breeding season (Stowe 1982, Murphy et al. 1986, Boekelheide et al. 1990). Reduced numbers at colonies during such phenomena result from absence, not mortality, of breeding adults; therefore, increases to former numbers generally occur within 1 or 2 years after such events (Birkhead and Hudson 1977, Stowe 1982, Boekelheide et al. 1990). Reductions in numbers of murres at colonies within the trajectory of the T/V Exxon Valdez oil spill have persisted for 3 years; thus, we think it is unlikely that murres were only temporarily away from colonies. Furthermore, a perturbation other than the spill sufficient in magnitude to affect colonies over such a wide area, from the Chiswells to Ugaiushak, should have similarly affected Middleton and the Semidi islands.

Besides reduced numbers of birds

following the spill, murre nesting behavior at colonies within the trajectory was disrupted. Nesting phenology (timing) was delayed and reproductive success was lower than normal. These effects have persisted for at least three years at monitored sites. Prior to the spill, murres at colonies in the western Gulf of Alaska typically initiated egg laying in June, but after the spill, the onset of laying was delayed until mid- to late July at the Barren Islands and Puale Bay, the two colonies for which we had data. No such delays occurred at colonies just outside the trajectory. Delays, and indeed failures to lay eggs, in 1989 could have been due to the loss of breeding birds, hydrocarbon contamination, food web disruptions, frequent disturbances due to spill cleanup activities or a combination of these factors. By 1990 oil was apparently no longer present near breeding colonies, and the level of human activity had diminished. Therefore, probable causes of disruptions to murres in 1989 were no longer a factor.

Persistent delays in nesting could have resulted from the abrupt declines in breeding populations with commensurate reductions in densities at most breeding ledges. Reduced densities could have caused social disruption at colonies. Social stimulation apparently is an important factor in the timing of laying in murres (Birkhead 1985) because murres within clusters tend to lay more synchronously than the colony as a whole (Birkhead 1977, Birkhead 1980, Harris and Wanless 1988, Schauer 1991). A critical density of murres on nesting ledges may be necessary to stimulate ovulation. Clusters of potential breeders may not have reached adequate densities until the arrival of young birds prospecting for nest sites, an event that nor-

mally happens after incubation is underway (Tuck 1961).

Most of the murres killed near colonies in 1989 were probably experienced breeders, because younger birds do not return to colonies until later in the season if at all (Birkhead 1977, Stowe 1982). The removal of many of the experienced birds probably resulted in a population containing a much higher proportion of young, inexperienced breeders than normal. We speculated that surviving experienced breeders had a high probability of pairing with inexperienced birds. Young birds lay relatively late, even under normal conditions (Perrins 1970, Birkhead and Nettleship 1981, Gaston 1991, Nobel 1991), so a skewed age distribution could have caused a delay in the onset of laying.

Drastic changes in neighbors at nesting cliffs could also have caused disruption of normal nesting behavior. Murres occupy the same nest ledges annually (Hedgren 1980); thus, a breeding concentration of murres would be composed of a high proportion of birds that had spent previous summers on the same ledge with each other.

Murres experienced nearly total reproductive failures at every colony we monitored within the trajectory of the oil following the spill in 1989. Success remained below normal in 1990 and 1991 (i.e., in most cases <10 chicks per 100 adults compared to the normal rate of >50). Persistently low reproductive success following the oil spill, like phenology, may have been due to social disruption, reduced densities and skewed age ratios. Murre reproductive success is positively correlated with the density of nests (Birkhead 1977, Hatchwell and Birkhead 1991), and densities must have been lower at all colonies with reduced

populations after the spill. Birkhead (1977) found that a decline of common murre populations reduced the density of breeding groups and exposed the eggs and chicks of the remaining birds to gull predation. Furthermore, young murres are usually less successful than older birds, thus a colony with a high percentage of inexperienced breeders would be expected to have low productivity (Hedgren 1980, Gaston 1991, Nobel 1991).

Delayed nesting phenology seemed to be partially responsible for chick mortality at Puale Bay, and probably contributed to reduced reproductive success elsewhere. For murres there is usually a seasonal decline in reproductive success (i.e., late laying results in poor success) (Birkhead and Nettleship 1981, Gaston et al. 1983, Birkhead and Nettleship 1982, Boekelheide et al. 1990), therefore delayed phenology would also contribute to lower productivity of murres.

Another possible cause of abnormal laying phenology and reproductive performance is food shortages near the breeding colonies. In our study area, this is an unlikely explanation, because nearby colonies outside the trajectory of the oil were not affected. Furthermore, at the Barren Islands in 1990, tufted puffins (*Fratercula cirrhata*) experienced a normal rate of reproductive success and chicks grew at normal rates (D. Boersma, Univ. of Washington, pers. comm.). Puffins are diving fish-eaters like common murres, so that if food were limiting, we would have expected puffins to have demonstrated poor success.

If social disruption and a skewed age distribution were causing reduced reproductive rates, productivity should have begun to increase as birds became more experienced. Indeed, success rates were slightly higher at the Barren Islands

and substantially higher at Puale Bay in 1991 than in 1989 or 1990. Nevertheless, it is too soon to know whether this trend will continue. Most likely, a return to more normal laying dates will have to precede a sustained improvement in reproductive success.

We conclude that murre populations at breeding colonies within the trajectory of the oil declined following the spill. Furthermore, surviving murre populations likely had disrupted social structures due to a preponderance of young, inexperienced breeders and reduced densities on nesting ledges. By 1991, we found neither an indication of increases in populations nor a return to normal laying phenology, but reproductive success was slightly higher than in previous post-spill seasons.

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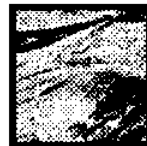
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How Long to Recovery for Murre Populations, and Will Some Colonies Fail to Make the Comeback?

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By anyone's estimate, hundreds of thousands of marine birds died in 1989 as a result of contact with oil spilled from the ruptured hull of the *Exxon Valdez*. One post-spill, computer-modelling project, funded by the Department of Justice, used data from a large study of the fates of radio-tagged bird carcasses to estimate the number of birds killed based on the number of carcasses recovered after the spill (Ford et al., 1991). That study concluded that the most likely estimate of the number of birds killed was 375-435 thousand individuals. They also estimated that there was only a 5% chance that fewer than 300 thousand birds died, and, more ominously, that there was nearly a 10% possibility that more than 600 thousand birds were killed. Based on the carcasses recovered on beaches and at sea, well over half of those birds must have been murres, and, given the relative abundance of the two murre species in the northern Gulf of Alaska, most of those must have been common murres (*Uria aalge*).

This was an unprecedented catastrophe for the common murres of the northern Gulf of Alaska. Data from the U.S. Fish and Wildlife Service's seabird colony catalog (Sowls et al., 1978) indicate that prior to 1989 only 230,000-350,000 breeding-age murres occupied colonies in those areas of the Gulf of Alaska most exposed to oil. Assuming that the population was at equilibrium prior to the spill, and that survival rates of different age classes were similar to those in populations for which

survival estimates have been obtained, the total population of adults and sub-adults would have been roughly 350,000-750,000 murres. Thus, it is quite possible that a large portion of the murres living in the region were killed by the *Exxon Valdez* oil spill.

Earlier research on the potential impacts of oil spills on Alaskan murre populations concluded that mortality of breeding adults would have the strongest long-term effect on the population (Ford et al., 1982). U. S. Fish and Wildlife research conducted since the spill has shown that at the two largest of the oil-impacted colony-clusters (Barren Islands and Puale Bay), the numbers of breeding murres that occupied their colonies in 1989 were down from previous years by 50-70% (Nysewander, 1990). Perhaps even more significantly, there has been very little rebound in subsequent years (Nysewander and Dipple, 1990; Dipple and Nysewander, 1991, Nysewander pers. comm.), suggesting that early recruitment of sub-adults to the breeding population or immigration from unaffected areas, processes that have accelerated recovery of seabird populations in other cases, are not as yet contributing to recovery. These data imply that on order of 120-140 thousand adult common murres must have died in the aftermath of the *Exxon Valdez* oil spill, and that recovery has not yet begun or is progressing very slowly.

Murres are long-lived birds with low

reproductive rates. So it is not surprising that our earlier investigation of long-term oil-spill impacts, Ford et al. (1982), also demonstrated that relatively small chronic decreases in fecundity rates could have a large impact on the ability of a population to recovery from oil-spill mortality. Indeed, it does not take very large declines before the population may be incapable of recovery. Unfortunately, the effect of the *Exxon Valdez* oil spill on the fecundity rates of murres has not been a slight decline, instead it has been reduction to nearly zero. Egg laying was delayed by a month in 1989 in the Barrens and at Puale Bay, and it is highly unlikely that any young were fledged that year (Nysewander 1990). This pattern was repeated in 1990 and 1991 (Nysewander and Dipple 1990, Nysewander pers. comm.). It appears that the density of murres on the breeding ledges has been reduced to such an extent that the early, synchronous egg laying that is crucial to successful reproduction in murres could not be achieved. It is likely that this density effect is a threshold phenomenon, which means that until densities climb above the threshold, reproductive rates will stay very low. This raises the possibility that some colonies may not recover at all, or will require very long times and the contribution of immigration from other areas to do so.

Studies of other populations of common murres suggest that the maximum population growth rate is approximately 4% per year. If the northern Gulf of Alaska population could achieve that rate and that only half the population perished in 1989, a best case scenario, then it would take approximately 18 years for the population to return to its pre-spill size. On the other hand, if the per-

centage killed was 70% and the growth rate is 2.5%, a more realistic scenario, it would take nearly 50 years for the population to recover! Of course, the density-threshold effect may be delaying the onset of any recovery indefinitely.

What does the future hold for common murres? Will all of the colonies recover, or will some become extinct? How long will it be before the seabird cliffs of the northern Gulf of Alaska again teem with murres?

I will describe computer simulation models built to answer these questions. Because of the uncertainties introduced by the density-threshold effect and the lack of any data on the pre- or post-spill survival rates of Alaskan murres, model predictions cover a wide range of possible outcomes. Nonetheless, early model results suggest that the recovery will most likely require several decades, and that some colonies or sub-colonies may face extinction. The models indicate that other poorly known factors, such as immigration and recruitment from other colonies, the response of predators and the occurrence of new oil spills can have a significant impact on the outcome of murre population recovery. While models such as these can be used to give us quantitative estimates of the likelihood of the various possible recovery scenarios for these murre populations, they may serve a greater purpose by helping to assess the potential effect of different population or colony restoration options. In addition, sensitivity analyses of the models can be used to point to the areas that need the most study to enable an effective monitoring of the recovery process and improved damage assessment and restoration effort the next time seabird populations suffer such a catastrophic oil spill.

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Effects of the Exxon Valdez Oil Spill on Bald Eagles

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In March, 1989, the tanker *Exxon Valdez* ran aground and spilled more than 11 million gallons of crude oil, fouling shorelines from Prince William Sound to the Alaska Peninsula. About 8000 bald eagles inhabit that area. A 3-year study was initiated soon after to assess damages to bald eagles. Specific objectives were to determine effects of the spill on bald eagle reproduction and survival of adults and fledglings, conduct population surveys to assess population response, and examine eggs, prey, and blood for evidence of hydrocarbon exposure. The greatest damages to bald eagles occurred in 1989 and were manifested by direct mortality of bald eagles throughout the spill area, and significantly reduced reproduction in Prince William Sound. Damages after 1989 were not documented.

Although 151 dead eagles were recorded at wildlife collection centers, these probably represented a small fraction of the total mortality. We adjusted this figure for carcasses not found, not reported, scavenged, drifted out to sea, or otherwise lost, and estimated that between 614 and 1871 eagles (best estimate was 902; about 11% of the population) died as a result of the spill.

Reproductive failures were directly related to the extent and intensity of shoreline oiling near nests, but effects on reproduction apparently did not extend beyond Prince William Sound. It was not possible to differentiate the effects of the oil itself from that of disturbance caused by shoreline cleanup operations.

The lack of observed reproductive failure in other areas was likely due to the later arrival of oil during the nesting season, a decreased toxicity due to weathering of oil, or changes in consistency of the crude oil which made oil less adherent as the slick moved westward along the coast. Bald eagle reproduction in Prince William Sound rebounded in 1990 to levels typical of other bald eagle populations in coastal Alaska.

Stratified random plot surveys of adult bald eagles within Prince William Sound provided indices of population size. Indices for 1982, 1989, 1990, and 1991 were 1565 ± 473 , 2089 ± 308 , 1941 ± 283 , and 2088 ± 273 , respectively. These indices represented approximately 47% of the total population size because they did not include immature eagles or account for adult eagles not seen during surveys. Population size did not change significantly from 1989 to 1990 or 1991. However, because confidence limits on estimates ranged from 13-15% for these 3 years, it is unlikely that we could have detected a statistically significant difference given the estimated magnitude of mortality caused by the spill.

Analysis of addled eggs and prey remains confirmed exposure to petrogenic hydrocarbons. However, the high incidence of exposed samples from eastern Prince William Sound in 1990 muddle interpretation and suggest that contamination from other sources may occur in areas distant from the spill area. Uric acid and other blood chemistry parameters were compared for birds

sampled in oiled and unoiled areas of Prince William Sound in 1989 and 1991. Although uric acid values were higher for eagles from oiled areas of west Prince William Sound in 1989, they were lower for birds sampled in the same areas in 1991, as compared to eagles sampled in east Prince William Sound. Because uric acid values were strongly influenced by the presence of food in the crop, we concluded that uric acid was not a reliable means of assessing chronic damages from oil. Concentrations of hydrocarbons in blood seemed low, but there is no information available to enable interpretation of these levels.

Survival was high for eagles radiotagged in Prince William Sound 4-5 months after the spill. We used three schemes to compare survival of birds radiotagged in oiled and unoiled areas. These schemes were based on (1) whether the eagle was tagged in east or west Prince William Sound, (2) presence or

absence of oil at the capture site, and (3) the percentage of relocations in oiled areas after tagging. There were no differences in survival between groups using any of the 3 schemes. Any adverse effect on survival likely occurred before we tagged the eagles. A population model of bald eagles in Prince William Sound indicates that the population was increasing before the spill at a rate of about 2% per year. This is consistent with the observed population increase from 1982 to 1989.

We believe that the effects of the oil spill were short term, and were limited to direct mortality throughout the spill areas, and impaired reproduction in Prince William Sound. The population model indicates that the cumulative effects of these damages will set back the eagle population in Prince William Sound by about 4 years, but population recovery and growth should occur naturally in the absence of other major disturbances.



Effects of the Exxon Valdez Oil Spill on Pigeon Guillemots (*Cepphus columba*) in Prince William Sound, Alaska

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The supertanker *Exxon Valdez* ran aground in Prince William Sound, Alaska, on March 24, 1989, spilling 260,000 barrels of Prudhoe Bay crude oil. We studied the effects of the spill on the population and reproduction of the pigeon guillemot (*Cepphus columba*), a diving seabird, at Naked Island, Prince William Sound, from 1989 to 1992.

The pigeon guillemot breeds ubiquitously (existing everywhere) on rocky shores along the west coast of North America, nesting in pre-existing cavities in coastal cliffs and talus. Guillemots feed in inshore waters on benthic (bottom) fishes (e.g., blennies, sculpins, cods), schooling fishes (e.g., sand lance, herring), and invertebrates such as crab and shrimp. Unlike most seabirds, guillemots nest in many, small colonies, rather than in a few, large colonies. This nesting distribution is thought to be related to their use of inshore feeding areas.

Naked Island, a major breeding site for guillemots in the Sound, is located 30 km southwest of the grounding site, and oil was observed on the waters surrounding Naked Island between March 29 and April 19, coinciding with the guillemot pre-breeding season. Oil remained on the beaches of certain Naked Island guillemot colony sites throughout the 1989 breeding season. Guillemots congregate on beaches at their colonies during the breeding season, and guillemots could therefore have been exposed to oil during their breeding activities.

To determine whether the spill

affected Naked Island guillemots, we compared reproductive data collected during the 1989 and 1990 breeding seasons and population data collected during 1989-1992 to data collected using similar methods during the 1978-1981 breeding seasons (Oakley and Kuletz 1979; Oakley 1981; Eldridge and Kuletz 1980; Kuletz 1981, 1983). We also collected a small sample of guillemots and eggs in 1989 and 1990 for analysis of petroleum hydrocarbon residues.

The Naked Island area guillemot population was 25-36% lower in 1989 than in the early 1980s, and the population continued to decline through 1992. In 1992, the population was only 1,016 guillemots; in 1978 and 1979, the population had been 1,965 and 2,230 guillemots. Thus, by 1992, the Naked Island population was roughly half its former size.

The Prince William Sound guillemot population declined by as much as 50% between 1972 and 1985 (Laing and Klosiewski in prep.), and the extent to which the decline we observed at Naked Island was due to this overall population decline or to the oil spill is unknown. However, the most heavily oiled areas on Naked Island were the areas with the largest declines in the number of guillemots in 1989. A similar result was found by Laing and Klosiewski (in prep.) for the Prince William Sound guillemot population as a whole: the guillemot population declined by 50% between 1985 and 1989, and the decline was twice as great in oiled areas as in unoiled areas.

The reproduction of guillemots nesting at Naked Island in 1989 was similar to previous years. Chronology was identical to that in other good weather years. Adults fed their chicks at similar rates, and chicks fledged at similar weights. Based on a small number of nests, hatching, fledging and nesting success rates were not significantly different than in pre-spill years.

Based on oil dosing studies (e.g., National Research Council 1985, Nero and Associates, Inc. 1987, Peakall et al. 1980), oil from the *Exxon Valdez* was most likely to have disrupted guillemot breeding by delaying egg laying, increasing egg mortality, decreasing hatching success, increasing nest abandonment and slowing the growth of chicks.

The relative normality of reproduction in 1989 suggested that the oil spill did not have a negative effect on guillemot reproduction. However, the cryptic nature of guillemot nests decreased our ability to detect one of the more likely effects the spill could have had: an increased incidence of unhatched eggs.

The number of active nests at five intensively-studied colonies increased from 21 nests in 1989, to 34 nests in 1990, while the maximum counts of attending guillemots did not change. This increase in the number of active nests suggested that breeding in 1989 was disrupted either through decreased hatching success or because fewer pairs initiated nests.

Oil was found on the surface of guillemot eggs which failed to hatch in 1989 and 1990, suggesting that the predicted effect of decreased hatching success occurred. The presence of oil on unhatched eggs during 1990 suggested that even though most Naked Island area beaches had been cleaned, guillemots were still being exposed to oil one year

after the spill.

Although the number of active nests increased in 1990, reproductive success was poor. Many nests failed due to low hatching success and predation. The percentage of schooling fish in chick diets decreased, and chicks grew at slower rates. The relationship between poor reproduction in 1990 and the oil spill is unknown.

Local guillemot population declines in Norway, Scotland, Denmark and California have been attributed to oil pollution. In most cases the populations appear to have recovered fairly quickly (Ainley and Lewis 1974, Heubeck and Richardson 1980, Nettleship and Evans 1985). The relatively large clutch of the pigeon guillemot, typically two eggs, gives guillemots the potential to rebuild their populations faster than most other alcid species, which generally lay only one egg.

Identification and protection of important nesting and feeding areas would facilitate restoration of guillemot populations affected by the *Exxon Valdez* oil spill. Knowledge of the distribution of guillemot colonies in the spill area is incomplete, because surveys specifically for guillemots have not been conducted, except at Naked Island. Guillemots nest at many sites where no other seabirds nest, and normal colony census practices only document guillemots breeding at sites where other seabirds also breed.

An understanding of the causes of the long-term decline in the Prince William Sound guillemot population is required to properly plan any guillemot restoration action.

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Quantification of Habitats in Prince William Sound from Landsat Thematic Mapper Satellite Imagery

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GAIA Software



The first goal of this project was to identify and quantify habitats in Western Prince William Sound, Alaska from Landsat Thematic Mapper satellite imagery. Secondly the goal was to assess the feasibility of using satellite imagery and other remote sensing products to measure the habitat of Marbled Murrelets impacted by the *Exxon Valdez* oil spill. Here I present detailed habitat information on a total of two million acres in the *Exxon Valdez* oil spill area including acreage reports of 18 surface features for the mainland and the eleven principle islands of western Prince William Sound. I also present full-color thematic maps for each of these geographic areas as well as a parcel analysis of forested slopes near streams, a habitat thought to be important to Marbled Murrelets and other *Exxon Valdez* oil spill impacted species. The Landsat image analyzed was acquired on August 13, 1986.

This work has important implications because satellite imagery represents an ideal source of data for mapping habitats over vast wilderness areas such as Prince William Sound. Because each satellite image is a single large data set collected at a single moment, a modest investment in ground-truthing a small area, such as a few thousand acres, allows one to extrapolate the results with a high level of confidence to the millions of acres contained in the entire image. In rugged coastal areas such as Prince William

Sound (where field seasons are short, logistics are expensive and operations are dangerous), remotely sensed data provide efficient coverage of vast and inaccessible areas, thereby reducing the time, cost, and risk associated with alternative methods of field survey.

In the entire study area, forests were the most abundant habitat, covering 28 percent of the region. Muskegs were the second most abundant habitat, covering 20.4 percent of the land area, and snow fields and glaciers were third at 13 percent. Rock, cloud, alpine and shrub thickets and several other habitat types each covered under 10 percent of the study area.

On islands, muskegs covered 25 percent of the land, whereas on the mainland they covered only 14 percent. Forests covered 34 percent of the islands compared to 19 percent on the mainland. Twenty-nine percent of the mainland was covered by ice fields and glaciers compared to less than 2 percent for this cover type on the islands. Sixty-one thousand acres of forested slopes (17%), a habitat possibly important to Marbled Murrelets and other *Exxon Valdez* oil spill damaged resources, were found on islands, as compared to 24,208 acres on the mainland (9 percent). An assessment of the accuracy of the thematic map produced from the Landsat data when compared to random points on aerial photographs yielded an accuracy of 91 percent.



Effects of the Exxon Valdez Oil Spill on Marbled Murrelets

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The March 24, 1989 Exxon Valdez oil spill caused immediate mortality to hundreds of thousands of seabirds. In Prince William Sound, where the grounding of the supertanker occurred, marbled murrelets (*Brachyramphus marmoratus*) were 12% of the recovered carcasses, although they were only about 6% of the seabird population at the time of the spill (Piatt et al. 1990). In the summer the marbled murrelet, a small diving alcid, is the most abundant seabird in Prince William Sound, and is common throughout the spill zone. Prince William Sound and the Kodiak Archipelago are believed to have a high percentage of the marbled murrelet population occupying U.S. waters (Mendenhall 1992). The marbled murrelet is a species of concern in Alaska and is listed as threatened under the Endangered Species Act in Washington, Oregon and California. Following the Exxon Valdez oil spill, I conducted an injury assessment study on the marbled murrelet in 1989 and 1990. Additional data were collected in 1991 and 1992 during related restoration studies.

The objectives of this study were to test for differences in murrelet numbers before and after the spill at selected sites, provide an index of breeding activity and test for exposure to petroleum hydrocarbons in adult marbled murrelets. In addition, I synthesized data on murrelet carcasses recovered following the spill and on murrelet populations from other oil spill studies.

To test for changes in local murrelet

populations, I compared pre- and post-oil murrelet counts on repeated transects at Naked Island, located in the center of Prince William Sound (1978-1980 v. 1989-1990) and Kachemak Bay in lower Cook Inlet (1988 v. 1989). Complete shoreline surveys were also available for Naked, Storey and Peak islands (1978-1980 v. 1989-1992). As an index of breeding success, observations of juvenile murrelets on the water at these sites during transects and shoreline surveys between mid-July and mid-August were compared between pre- and post-oil years. To test for petroleum hydrocarbon contamination, 28 birds were collected from three locations in Prince William Sound (un-oiled, lightly oiled and heavily oiled sites).

On five mid-bay transects at Naked Island there were significantly fewer murrelets in 1989 compared to 1978-1980, but the number of murrelets on these same transects in 1990 were comparable to pre-spill numbers. Shoreline murrelet counts of Naked, Storey and Peak islands were also significantly lower in 1989, at less than 30% of pre-spill counts. However, the mean of murrelet counts from 1989-1992 was not significantly different from pre-spill means, although the post-spill mean for Naked Island was only 68% of the pre-spill mean. In Kachemak Bay, I found no significant difference in counts on transects between 1988 and 1989. At both study sites, there was a negative relationship between boat or aircraft activity and murrelet numbers.

During transects and shoreline surveys at Naked Island, significantly fewer juveniles were counted on the water in post-oil surveys than were counted in pre-oil surveys. There was no significant difference in juvenile counts at Kachemak Bay between 1988 and 1989. Birds collected in Prince William Sound in 1989 in oiled areas showed petroleum residue in liver tissue, whereas none of the birds collected at the unoiled site were contaminated.

I concluded that in the Naked Island complex, a moderately oiled area, the low numbers of murrelets in nearshore waters in 1989 was due to disruption of breeding activity by human disturbance that year and partly to mortality from the spill. Based on juvenile counts on the water, reproductive success was low in post-oil years, particularly in 1989. The occurrence of liver contamination in adults which appeared healthy suggested that murrelets were contaminated by eating contaminated prey, which could have long-term effects on the population. There was no evidence of population declines or lower reproductive success for murrelets at Kachemak Bay, which was only lightly oiled with weathered crude oil near the mouth of the bay.

The July murrelet population of Prince William Sound has declined 67% since the early 1970's (Laing and Klosiewski, in prep), but the contribution of the spill to their decline is uncertain. Most murrelet carcasses were retrieved from the lower southwest islands of Prince William Sound before mid-April, when many murrelets were migrating into the Sound. The dispersal of birds to their breeding sites once they migrated into the Sound, would have masked any strong association between low numbers of birds and oiled areas.

At the time of the spill, marbled murrelets were probably beginning to move into nearshore waters of southcentral Alaska, but the majority of murrelets would not have reached their summer breeding locations until early May. This study found that marbled murrelet numbers increased throughout April in Kachemak Bay and reached summer concentrations in mid-May. Along the southern Kenai Peninsula, Vequist and Nishimoto (1990) documented a 30% increase in murrelets between surveys in early and late April, and another 78% increase by June. The increase in murrelet numbers coincides with their expected movement into breeding areas.

Murrelets already in the southwest corner of Prince William Sound or along the southern Kenai Peninsula would have been affected by the spill throughout April, as the oil moved south. Applying the model derived by Ford et al. (1991) to carcass counts, I estimated the immediate mortality of *Brachyramphus* murrelets from the spill to be between 10,200 - 22,000 murrelets, with a best approximation of 12,700 - 14,800 murrelets. Because of their small size and low probability of carcass beaching and recovery, especially outside the Sound, I consider this a minimum estimate of direct mortality from the oil spill.

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Acute and Sublethal Effects of the Exxon Valdez Oil Spill on Harlequins and Other Seaducks

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This paper describes effects of the Exxon Valdez oil spill on seaducks. The study found that effects are linked to two causes: oiling, both sublethal and direct, and disturbance. Comparisons are presented for reproductive rates of Harlequin Ducks in oiled and non-oiled areas.

Six species of seaducks commonly occur in the Exxon Valdez oil spill area: these are Harlequin Ducks, Barrow's Goldeneyes, Common Goldeneyes, Surf Scoters, Black Scoters, and White-winged Scoters. These seaducks forage mainly in invertebrate prey in different areas: Harlequin Ducks forage mainly in the upper and middle intertidal zone on small clams, chitons, limpets, hermit crabs and blue mussels; goldeneyes and Surf Scoters feed in the lower intertidal and shallow subtidal zones on larger blue mussels and snails; White-winged Scoters forage in deeper water (up to 30 m) on scallops and clams.

The Harlequin Ducks became the main focus of this study after the spill because of their particular susceptibility. Harlequin Ducks have a population in the affected area that consists of both resident and migratory birds. The residents breed in the spring along forested streams within a few kilometers of saltwater; the hens raise young ducklings on streams but move them to saltwater later in the summer. All adults molt in secluded bays and lagoons in late summer. When not occupying nests, Harlequin Ducks can often be seen roosting on offshore rocks.

Non-resident Harlequin Ducks spending the winter in Prince William Sound breed elsewhere in Alaska on mountain streams. They arrive in the southcentral area in October and depart in May. The summer population of resident Harlequin Ducks in the Prince William Sound oil spill area was estimated at approximately 2000 individuals.

All six species of seaducks were in the path of the spill and suffered varying degrees of acute mortality. In addition, by virtue of their nearshore habitats, they were chronically exposed by oil remaining in the intertidal by direct contact to feathers and skin, and internally through preening and ingestion of contaminated food (e.g., in blue mussels). This was especially true of Harlequin Ducks which forage in the upper intertidal zone where the greatest amounts of oil remained after the spill. In addition to direct oil exposure, Harlequin Ducks also were effected by the massive clean-up effort of many beaches in Prince William Sound. These activities may have temporarily displaced or disturbed the normal reproductive cycle of this species.

Information on acute mortality came from counts of birds in the morgues (Piatt, 1989), and from comparing pre-spill and post-spill population counts from boat surveys conducted by the U.S. Fish and Wildlife Service (see Klosiewski and Laing, this volume).

The post-spill studies were carried out to determine if there were sublethal effects to seaducks, particularly resident

Harlequin Ducks in Prince William Sound. For these studies the oil-exposed ducks in western Prince William Sound were compared to unexposed populations in eastern Prince William Sound. Internal exposure was determined by analysis of liver for petroleum hydrocarbons and bile for hydrocarbon metabolites. Body condition and fat content were determined by examination. Breeding activity was determined by mist-net collection efforts at the mouths of streams with potential nesting habitat for Harlequin Ducks. Reproductive success in the two portions of Prince William Sound was determined by a census of the Sound shoreline during the late summer for hens with fledged ducklings.

Collections of Harlequin Ducks were also carried out during the late-summer molt period of harlequins when they could be captured live. The ducks were released after taking blood samples for biochemical analysis for oil-induced steroid changes and fecal samples for hydrocarbon analysis.

There was significant immediate mortality of seaducks from the spill. Up to September 25, 1989, 148 carcasses of Harlequin Ducks arrived at the U.S. Fish and Wildlife Service Valdez receiving station. Applying the estimated 35% recovery rate of carcasses to this figure resulted in an estimated total acute loss of 423 Harlequin Ducks in Prince William Sound.

For the other species total recovered carcasses from all receiving stations in the spill areas are as follows: 33 Barrow's Goldeneyes; 6 Common Goldeneyes; 25 unidentified goldeneyes; 175; Surf Scoters; 132 Black Scoters; 342 White-winged Scoters; 162 unidentified scoters. No estimates of total mortality and the proportion of the population in the spill

areas that this represents have been made for these other species.

Harlequin Ducks collected from western Prince William Sound had indications of elevated exposure internally to petroleum hydrocarbons compared to those collected in eastern Prince William Sound in 1989. The mean concentration, for example, of compounds in bile samples fluorescing at phenanthrene wavelengths were 56,000 ug g⁻¹ in western Prince William Sound and 10,030 ug g⁻¹ in the control area.

Comparison of body condition of Harlequin Ducks indicate that those in eastern Prince William Sound were in much better condition than those in the western Sound.

Mist netting efforts to capture birds attending nests on streams in 1991 and 1992 were as follows: in 1991 in eastern Prince William Sound 12 streams were netted for 149.5 hours and 23 ducks were captured, banded and released. Fourteen breeding hens were radio-tagged and five nests were located. In 1992 in eastern Prince William Sound 20 streams were netted for 485 hours and 44 ducks were captured. Thirty-two breeding hens were radio-tagged. Three new nests were located.

In 1991 in the western Prince William Sound oil spill area 16 streams were netted for 132 hours and 0 ducks were captured. In 1992, 37 streams were netted for 254 hours. Only 2 ducks were captured at Hanning Bay on Montague Island on the extreme periphery of the oil spill area. These two hens were radio-tagged. One nest was located by radio-tracking the hen to a site on Hanning Creek.

Based on mist-netting and observations of pairs at stream mouths, we estimate that there are an average of 5 pairs

of ducks per nesting stream in eastern Prince William Sound and 0 pairs of ducks per stream in the western Sound oil spill area. The data indicate significant differences in the distribution of breeding pairs in the oiled and non-oiled study areas.

The results of late summer shoreline census for harlequin hens with ducklings in 1991 and 1992 were as follows: in eastern Prince William Sound in 1991, 232 miles of shoreline searched; 16 hens were sighted with an average brood of 3.4 ducklings. In 1992, 248 miles of shoreline were searched; 5 hens were sighted with an average brood size of 4.0 ducklings. A cold, late spring probably reduced harlequin productivity in eastern Prince William Sound in 1992 as it did for many other waterfowl species in Alaska.

During 1991, 285 miles of shoreline in the western Prince William Sound oil spill area and periphery were searched. Results of the shoreline survey in the oil spill area per se were the following: a single harlequin hen with a late brood of three ducklings was sighted in mid-September in Bay of Isles, Knight Island. One hen with three ducklings was observed at Johnson Bay on the west side of Knight Island on the periphery of the oil spill area. One hen with a brood of four was observed at Whale Bay in southwestern Prince William Sound. Two other hens with broods of 3 and 4 ducklings were recorded at Hanning Bay, Montague Island. None of these bays were oiled.

In western Prince William Sound in 1992, 465.7 miles of shoreline were surveyed in the oil spill area and periphery. One brood of five was sighted on the west side of Knight Island near Drier Bay; and one brood of three was sighted in Hanning Bay. No other broods were sighted.

The prespill data indicate that Harlequin Ducks raised ducklings successfully throughout western Prince William Sound (Isleib and Kessel, 1973; Oakley and Kuletz, 1979; Dzinbal, 1982; Isleib, personal communication). The data collected after the spill indicate that there was a massive reproductive failure in Harlequin Ducks.

There appear to be two potential causes for the reproductive failure of Harlequin Ducks. One is that oil exposure from contaminated intertidal food items ingested by the ducks caused cessation of reproduction. The other potential cause is the effects of human disturbance from the massive clean-up of contaminated shorelines from 1989 through 1991. Since the reproductive failure continued in 1992 and disturbance levels declined each year and stopped in June 1992, disturbance is considered at most a secondary contributing factor. We favor the oil exposure hypothesis, i.e., the ducks are continuing to ingest contaminated food.

It has been clearly documented that the crude oil spilled from the *Exxon Valdez* exposed marine invertebrates such as blue mussels. We determined that as many as 45 blue mussel beds retained significant amounts of *Exxon Valdez* oil spill petroleum in western Prince William Sound in 1992. Blue mussels are a key prey species for seaducks. As a result of an investigation of the pathway of oil exposure, we have proposed that blue mussels from oiled mussel beds are an agent of transmission of petroleum hydrocarbons from the environment. The oil remains trapped beneath byssal thread mats in anoxic conditions and retains toxic components. Bioaccumulation in the food chain has resulted in uptake of petroleum hydrocarbons by Harlequin

Ducks over a long period. How long Harlequin Duck reproductive failure will continue is unknown, especially in consideration of the unweathered *Exxon Valdez* oil spill crude oil remaining beneath mussel beds. Previous studies have shown that very small single doses of petroleum exposure, either from ingestion or by preening oiled feathers, caused cessation of reproduction in certain seabirds for up to a year. Birds fed single doses of petroleum oils also exhibited altered yolk structure and reduced hatchability of eggs. Unless measures are taken to remove oil from mussel beds, it is possible that a local extinction of Harlequin Ducks may occur within the oil spill area.

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Black Oystercatchers in Prince William Sound: Oil Spill Effects on Reproduction and Behavior

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Reproductive success of Black Oystercatchers was studied in Prince William Sound at Green Island (oiled) and Montague Island (non-oiled) in 1989 and 1990; Brad Andres, now with U.S. Fish and Wildlife Service (USFWS) in Anchorage, obtained additional data at our study sites and at oiled and non-oiled sites at Knight Island in 1991. In 1989 fourteen oystercatcher breeding pairs on Green Island, three on Channel Island, and 20 on Montague Island were monitored from late May until mid-July. In 1991 the number of nests at Green Island was 21, an increase of seven pairs, among them five pairs that were present in 1989 but either did not attempt to breed or failed earlier in the season and did not attempt to re-nest. In 1989 mean hatching date of chicks in nests on Green Island was six days later than Montague Island (June 29 vs June 23) ($p=.10$). Nesting chronology was earlier at Green Island in 1990—most pairs checked had chicks by June 22—and was earlier at all Prince William Sound study areas in 1991 (mean hatching date June 18). In 1989 eggs on Green Island were smaller than on Montague ($p<.07$). Due to effects of oiling or disturbance by cleanup crews, black oystercatchers may have laid smaller eggs because a higher proportion of earlier clutches failed (second clutches tend to be smaller), or because they ingested oil which affected them physiologically.

Habitat parameters were measured

on meter-square quadrats from high to low tidelines in oystercatcher feeding areas in 1989. Oil was present on 91% of 53 transects, on 59% of quadrats within the transects, and mean percent oil cover was 23% on plots with measurable oil. Mussels collected within oystercatcher feeding territories at Green Island in 1989 were severely contaminated with Exxon Valdez hydrocarbons, and contained as much as 28 mg oil/g, or 0.3% of mussel body weight (Jeff Short, personal communication, Nov 1992), but were at minimum detectable concentrations on Montague Island. On Green Island, mortality of blue mussels was significantly higher (35% vs 16%, $p<.01$), mortality of *Fucus* was higher (78% vs 0%), and mussels tended to be smaller (33.9 vs 38.2 mm) than on Montague Island. In 1989 adult oystercatchers fed at a significantly slower rate on Green Island than on Montague (1.5 vs 4.2 ingestions per minute, $p<.05$), and their feeding bouts were longer (21 vs 11 minutes).

Hatching success, surprisingly, was higher at Green Island, the oiled site, in 1989 (73% vs 53% for Montague Island). Predators were responsible for destroying a larger proportion of nests on Montague Island, the non-oiled site. Oystercatcher hatching success in the oiled area may have been higher due to possibly reduced efficacy of mammalian predation on oil-contaminated shorelines in 1989. We were surprised that eggs were not oil-contaminated by incubat-

ing adults. It is possible that oystercatchers avoided oiled areas, but no data were collected in either 1989 or 1991 to determine whether oystercatchers were able to distinguish between oiled and non-oiled feeding areas, i.e., whether they fed in non-oiled areas disproportionately. In 1991, it was found by Andres that oystercatchers tended to remove non-oiled limpets experimentally presented to them, in preference to oiled limpets.

In 1989, daily chick mortality was directly correlated with degree of oiling, with 6% daily mortality on heavily oiled feeding territories, 3% on moderately oiled territories, and no mortality on the non-oiled control site. Fledging success was therefore lowered on Green Island in 1989. It should be noted that what Alaska Department of Environmental Conservation officials defined as lightly oiled had moderate brood reduction effects. Similarly, moderate oiling, as defined by ADEC, had severe brood reduction effects in our study.

In 1990 oiling was still pervasive on "cleaned" shorelines at Green Island, and fewer chicks fledged on heavily oiled than moderately oiled territories in 1990 (0.5 vs 1.1). On 15 intertidal transects on Green Island examined for oil in 1990, Mary Cody found oil on 41% of quadrats and on plots which had measurable oil, the average oil cover was 17% (vs 21% in 1989). On two of eight oystercatcher feeding territories, average oil cover was higher in 1990 than in 1989. ADEC data also indicate that oiling increased on 8 of 17 oystercatcher territories on Green Island from fall, 1989 to spring-summer, 1990. In the Exxon Valdez-affected area, the known extent of oiled shorelines increased from 1989 to 1990 on the Kenai Peninsula by 208% and on the Kodiak-Alaska Peninsula by 189%.

Observed brood reductions in 1989 may have been caused by the direct oiling of chicks, which are flightless for 35 days. There were two observations of directly oiled chicks in 1989. Chick mortality was probably also caused by the effects of ingesting oil in food. As noted above, mussels were heavily contaminated with hydrocarbons in 1989, and concentrations were probably toxic.

In 1990, Inipol was applied to Green Island to accelerate biodegradation of oil. No chicks fledged in bioremediated areas, compared to a small number fledged in areas that were oiled only. Hydrocarbon concentrations in mussels collected at Green Island in 1990 were close to minimum detectable concentrations. In 1991 Andres found that chicks in oiled areas gained significantly less weight than those on non-oiled sites. However, this effect was not translated into lower overall productivity in 1991.

Additional deaths of oystercatchers due to the Exxon spill are calculated from observations of the shoreline density of oystercatchers, surveys of oiled shorelines, and impacts of shoreline oiling on oystercatchers extrapolated from Prince William Sound study areas. ADEC estimates of oiled shorelines underestimated the degree and extent of oiling at Green Island. At 14 of 16 oystercatcher feeding territories on Green Island, we found significant oiling on meter-square quadrats, whereas maps generated from cumulative ADEC (overflight) data through August show oil at only 5. If fall 1989 Walkathon data are added to the cumulative map, the number of oiled territories increases to 8, though 2 territories were classified as very lightly oiled where we found moderate oiling during the summer. If both fall, 1989 and spring-summer, 1990 ADEC data are cumula-

tively added, oil is shown to be present on 13 of 16 oystercatcher territories, though by 1990 the ADEC oiling category for 8 of these is "very light" whereas we measured moderate to heavy oiling in summer, 1989, at those 8 sites.

Data from more intensive USFWS surveys of Becharof and Kodiak National Wildlife Refuge shorelines indicate that ADEC estimates of shoreline oiling were low for the Kodiak-Alaska Peninsula. For the Becharof coastline, ADEC shoreline oiling estimates from fall, 1989, were 7.1% of those conducted by refuge personnel the previous spring (12.7 km of oiled shoreline vs 178.2 km.). A survey of the Kodiak National Wildlife Refuge coastline by refuge personnel in the summer of 1989 provided an estimate of 3,146.3 km of shoreline oiled, compared to 122.8 km by ADEC in the fall for Alaska Peninsula-Kodiak Island.

The differences between ADEC data, ours, and USFWS surveys are probably due to the facts that (1) the available ADEC data from the Alaska Peninsula were collected in the fall; (2) USFWS examined more shoreline than ADEC personnel—only 247 km of 11457 km of the Kodiak-Alaska Peninsula shoreline were observed by ADEC; (3) in the fall, ADEC did not revisit some oiled beaches that were cleaned and officially "signed off" in the summer; (4) at the time of the arrival of oil on shore, many beaches surveyed from the air and appearing unoiled were in fact oiled when examined on the ground; and (5) on sandy beaches oil is quickly buried by sand deposition after its arrival.

The best available estimates of oiled shoreline are the cumulative totals of the aerial surveys conducted by ADEC in spring and summer for Prince William Sound and Kenai coasts, though as noted

they may differ from independent estimates. For Kodiak-Alaska Peninsula, USFWS we used refuge data. A total of 579, 288, and 3146 km of shoreline were estimated oiled by the *Exxon Valdez* in Prince William Sound, the Kenai Peninsula, and Kodiak-Alaska Peninsula Sectors, respectively, for a grand total of 4013 km. Of these, 201 km were heavily oiled, 282 moderately, 697 lightly, and 2834 very lightly. Hydrocarbon contamination of mussels has been documented throughout the *Exxon Valdez* oil spill-affected area.

Nine black oystercatchers were recovered by search crews from all sectors (Prince William Sound, Kenai Peninsula, Kodiak Island, and Alaska Peninsula) and were submitted to the morgue. Using a recovery rate correction of 7.5%, 120 adult oystercatchers may have been directly affected by the oil spill. Boat surveys conducted by USFWS found a significant reduction in the numbers of oystercatchers on oiled transects in Prince William Sound compared to non-oiled transects.

Lost production of chicks from these dead adult birds was estimated at 1290 chicks over the expected life of the adults.

Chick losses from possible breeding attempts in 1989 were calculated using an initial brood size of 1.87, which is derived from a mean clutch size of 2.56 times the mean hatching success of 73% observed for Green Island in 1989, and chick losses on lightly oiled and medium-heavily oiled (as defined by ADEC) of 3% and 6% per day, which result in overall chick losses of 66% and 89% at fledging age (35 days). No chick mortality to oil was assumed for shorelines classified by ADEC as very lightly oiled, though data collected from Green Island in 1989 indicated a loss of chicks on shore-

lines where oil cover was 7-10%, the upper limit of the ADEC very light category. Based on these assumptions, an estimated 635 chicks were not produced on shorelines in Prince William Sound, Kenai Peninsula, and Kodiak-Alaska Peninsula in 1989. Approximately 6 percent of the potential chick production along the Gulf of Alaska shoreline was probably lost in 1989.

In 1990, loss of chicks was still occurring at Green Island due to oiling and/or bioremediation. Indications are that bioremediation was used on 320 km of shoreline. Assuming a complete reproductive failure on bioremediated areas, that most bioremediation occurred in

Prince William Sound, and a breeding density of 0.49 pairs of black oystercatchers/km, 157 breeding pairs would not have raised 293 chicks in 1990. Insofar as continuing losses of chicks due to oiling in 1990 are concerned, it is safe to say only that on 82.7 km of heavily and moderately oiled shoreline in 1990, the minimum loss of chicks due to oiling, separate from bioremediation, was 19.

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Effects of the Exxon Valdez Oil Spill on Black-legged Kittiwakes in Prince William Sound.

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Black-legged kittiwakes are the most abundant colonially nesting seabird in Prince William Sound. Approximately 40,000 kittiwakes nest at 27 colonies in the Sound. The number of breeding pairs and reproductive success were monitored at 24 of these colonies from 1984 to 1992. Of these 24 colonies, 10 colonies or the area adjacent to them were oiled by the Exxon Valdez oil spill and 14 colonies were not oiled. The unoiled colonies were used as a control to determine if the oil spill had negative effects on the birds at the oiled colonies.

The number of breeding pairs did not decline at colonies in the oiled area after the Exxon Valdez oil spill when compared to the pre-spill years. Reproductive success of kittiwakes (number chicks fledged/nests built) in 1989 at the oiled colonies was about one half of what was expected based on previous years and the reproductive success of birds at the unoiled colonies ($P = 0.04$). From zero percent to 37 percent of birds at oiled colonies were observed during June or July of 1989 with oil on their breast feathers and no birds at unoiled colonies had oiled breast feathers.

Reproductive success of kittiwakes at all colonies in Prince William Sound declined in the post-spill years (1990, 1991, and 1992), compared to the 5 previous years. The brood size of fledglings also decreased in the post-spill years which suggests that there was less food available during these years (Irons 1992).

Results from contaminant analysis demonstrated that in 1989 one of ten birds from oiled colonies had livers that were contaminated by petroleum hydrocarbons and a single egg collected in the oiled area had a contaminated shell. In 1990 none of the five birds collected in the oiled area were contaminated, but two of the five had contaminated stomach contents. If this contamination resulted from the oil spill, it suggests that oil may have persisted at least a year in the food chain. Only 25 percent of the birds collected for contaminant analysis have been analyzed.

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Marine Bird Populations of Prince William Sound, Alaska, Before and After the Exxon Valdez Oil Spill

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We estimated the abundance of marine birds in Prince William Sound following the Exxon Valdez oil spill, examined population changes between pre-spill and post-spill surveys, and compared population trends in oiled zones of the Sound to trends in unoiled areas. Data from pre-oil spill boat-based surveys for birds in winter and summer 1972 and 1973 and a summer shoreline survey in 1984 were compared to data from this study collected in the winters and summers of 1989, 1990 and 1991 (Laing and Klosiewski, in prep.).

We counted approximately 100 bird species on surveys. Population estimates of 11 species or species groups declined between 1972/1973 and the years after the oil spill, including large declines for loons (*Gavia* spp.) (>36%), scoters (*Melanitta* spp.) (>54%), Arctic tern (*Sterna paradisaea*) (>78%) and murrelets (*Brachyramphus* spp.) (>65%). None of the species had population estimates increase significantly. Using one-tailed t-tests, we detected a net population loss ($p < 0.05$) in the oiled zone relative to population trends in the unoiled zone for pigeon guillemot (*Cepphus columba*) in March and northwestern crow (*Corvus caurinus*) in July, and marginally insignificant ($p < 0.10$) losses for cormorants (*Phalacrocorax* spp.), harlequin duck (*Histrionicus histrionicus*) and black oystercatcher (*Haematopus bachmani*).

In shoreline habitats, using the 1984 survey as a baseline, we estimated net loss in the oiled zone relative to the unoiled zone; if the 95% confidence in-

terval excluded zero, we concluded a loss occurred. Out of 18 species or groups examined in shoreline habitats, oiled zone losses were estimated to have occurred for loons, harlequin duck, scoters, black oystercatcher, Arctic tern, and mew gull (*Larus canus*). These effects were observed in 1990 and 1991, but not in 1989.

We conclude that oiled zone populations of nearshore species such as harlequin duck, black oystercatcher, pigeon guillemot and northwestern crow, as well as several offshore species, declined. Individual studies on harlequin duck, black oystercatcher and pigeon guillemot documented direct effects of oiling which may explain the population declines shown here (Andres et al., in prep.; Oakley and Kuletz, in prep.; Patton in prep.). In addition, overall population declines since 1973 of 11 species or species groups cause concern.

Statistically rigorous sampling design has rarely been used to estimate marine bird populations, and this study served to demonstrate its feasibility. However, the real value of using sampling to estimate populations is to illuminate long-term trends through repeated surveys. The lack of power of statistical tests in this study occurred because there were few baseline or post-spill surveys conducted, and because the baseline surveys occurred years before the oil spill. We hope that this study will provide scientists and policy makers with the impetus to survey populations at frequent intervals using rigorous sampling design.

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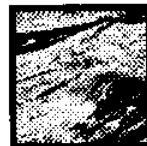
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Bioavailability of Residual PAHs From the Exxon Valdez Oil Spill

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Smith Island, located 25 miles southwest of the Bligh Reef grounding site, had one of the more heavily impacted shorelines from the Exxon Valdez spill. The coast of Smith Island is exposed and rocky, with boulder/cobble pocket beaches. Many of these beaches overlie beds of sand and gravel, and when oil came ashore it penetrated deeply into the substrate. Despite large-scale removal efforts through such means as high-pressure, hot-water washing, chemical agents, and excavation with heavy equipment, oil remains buried in portions of some beaches. At these locations, oil sheens have been observed to leach out from the substrate each year since the spill. The purpose of this study was to assess the extent to which residual polynuclear aromatic hydrocarbons (PAHs) were available for accumulation by intertidal organisms living at the site, and to evaluate the physical means by which organisms were exposed.

Since 1990, National Oceanic and Atmospheric Administration staff have sponsored a monitoring effort in Prince William Sound to evaluate the effects of both oiling and treatment at selected sites. An integral part of the program has been chemical hydrocarbon analysis of both sediments and tissues of intertidal invertebrates found at the sites. In 1990, mussels collected at the Smith Island site contained the highest concentration of PAHs of the 23 sites sampled for the National Oceanic and Atmospheric Administration Prince William Sound monitoring study, 84 parts per million (dry

weight). Mussel tissue samples from nearly all other sites contained less than 10 parts per million. The comparatively high value encountered for Smith Island mussels was not reflected in results of hydrocarbon analyses of surface sediment samples from the site, although very high concentrations of both total petroleum hydrocarbons and PAHs have been consistently found in subsurface sediments over the period between 1989 to 1992 (Michel and Hayes, 1991; Michel and Hayes, in preparation).

These conditions at the Smith Island site resulted in a more detailed study being implemented at that location for the 1991 field season. In addition to standard monitoring analyses of native mussels and composite sediment samples, as had been done in 1990, mussels from a nearby site in Eshamy Bay not impacted by the oil spill were collected and subsequently transplanted to two portions of the Smith Island study area. These were collected two months later and analyzed. The results suggested a rapid and substantial bioaccumulation of PAHs, with mussels increasing their body burden from 0.7 parts per million in the transplant stock, to 5 and 20 parts per million after two months. In contrast, mussels at the Eshamy Bay donor site showed a slight decrease in total hydrocarbon loading over the same period of time, to 0.3 parts per million.

In 1992, study activities at Smith Island were further expanded and refined. Bioavailability and transport of residual oil were evaluated through analysis of

PAH concentrations and distributions in several matrices, including mussels, an artificial mussel surrogate, sediment, water, and oil sheen. "Clean" mussels were collected from an unimpacted location in Prince William Sound (Barnes Cove, Drier Bay) and transplanted to an oiled site on the north side of Smith Island, as well as to a site with similar physical substrate characteristics on the relatively unimpacted south side. Most of the mussels were transplanted to the beaches in small cages placed directly on the pebble substrate underlying the large rounded cobbles, with cobbles replaced after deployment. Additionally, anchored and buoyed cages were located just offshore of the intertidal study area in an attempt to evaluate differences between intertidal and subtidal conditions. Half of the deployed mussels were collected after 14 days in the field, and the remaining half after 52 days.

A new monitoring tool developed by the National Fisheries Contaminant Research Center of the U.S. Fish and Wildlife Service was also deployed with the mussels. These simple devices—semipermeable membrane devices (SPMDs)—are essentially polyethylene envelopes containing triolein lipid, and are designed to act as bioaccumulation surrogates through passive uptake of lipophilic contaminants such as PAHs (Huckins et al., 1990). SPMDs, also referred to as "lipid bags", were paired with groups of mussels to evaluate route of exposure to the mussels, as SPMDs are thought to selectively sample the dissolved fraction of hydrocarbons. In a broader perspective, it was hoped that the experiment would permit some insight as to whether these devices might realistically be considered for monitoring effects of oil spills.

As was the case with mussels, SPMDs were placed in the middle intertidal zones of the target beaches as well as offshore from the beaches on buoyed deployments. In addition, to account for possible atmospheric contribution of PAHs to the SPMDs in the intertidal zone, SPMDs were placed in the supratidal berm where they were exposed only to air. Half of the deployed SPMDs were collected after 14 days in the field, and the remaining half after 52 days.

Some problems were encountered with physical stability of the SPMD and mussel deployments over the 52-day term of the study. Dynamic conditions on the beaches resulted in movement of some of the cages in which the mussels and SPMDs were housed, and loss of some SPMDs. However, stations were essentially intact for the 14-day recovery, and good results were obtained for those samples.

Analysis of the 14-day SPMDs showed a statistically significant uptake in lipid bags deployed in the intertidal zone of the oiled north side of the island, relative to the unoiled south side. Absolute levels of hydrocarbons accumulated in the intertidal SPMDs were, however, low relative to values measured in mussels in 1990 and 1991: the maximum summed concentration for target PAHs was about 1.4 parts per million. Interestingly, the highest accumulation of PAHs (about double that of the highest intertidal concentration measured) occurred in the SPMDs deployed on the supratidal berm on the north side of Smith Island, while supratidal SPMDs on the south side were only slightly elevated above blanks. Because it is known that SPMDs are also efficient air samplers, this suggests that volatilization of PAHs continues to be a pathway for loss of hydrocar-

bons from the buried residual oil that remains on the north side of the island.

PAH analyses were also performed on nearshore water, beach sediment in the immediate vicinity of the mussel and SPMD deployments, and on sheen observed leaching from the substrate on falling tides. This information was used to characterize PAHs in the Smith Island environment and as a basis for comparison to PAH profiles found in the mussels and SPMDs.

Chemical results were used to evaluate the extent to which PAHs had accumulated in mussels and SPMDs, and to give information on uptake rates. Patterns of PAHs in sediments, water, and visible sheen provided a basis for proposing a model of biological exposure.

We hope that by providing fundamental information on residual oil fate, bioavailability and mechanisms of exposure, this study may facilitate improved operational guidance on response and clean-up for future spills.

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Impact of the Exxon Valdez Oil Spill on Intertidal Invertebrates Throughout the Oil Spill Region

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The Coastal Habitat Injury Assessment Study was initiated to document and quantify injury to biological resources in shallow subtidal, intertidal and supratidal (immediately above the high tide level) habitats impacted by the Exxon Valdez oil spill. The coast was divided into three major regions: Prince William Sound (PWS), Cook Inlet-Kenai Peninsula (CIK), and Kodiak Island-Alaska Peninsula (KAP). This report deals with some of the impacts documented by the intertidal invertebrate component. Results for intertidal algae are presented separately.

The coastline in the above three regions was surveyed by the Alaska Dept. of Environmental Conservation to determine the degree of oiling relative to a GIS map of shoreline habitat types. The sites were classified as heavily, moderately, lightly or non-oiled; habitat classifications included sheltered rocky, coarse textured, exposed rocky, fine textured and estuarine. Oiled sites were randomly selected from each habitat type. A list of potential control sites for each oiled site was randomly generated and, through a ground truthing (verification) process, a control site was selected from the list as a matched site for each oiled site. Control sites included lightly oiled and non-oiled locations. Experimental sites included heavily and moderately oiled locations. This presentation deals only with sheltered rocky and coarse textured habitats.

At each site samples were collected along six transects (straight lines) perpendicular to the water line and evenly spaced along the beaches. Samples were collected within each of the first three meters of vertical drop (MVD) below mean higher high water along each transect. Quadrats (sampling squares), 0.1 m², were randomly positioned within each meter drop and all organisms within the quadrat were removed. New transect lines were established three meters to the left of the preceding transects on subsequent sampling visits; sites were sampled three to four times between fall of 1989 and summer of 1991. In the laboratory, organisms were sorted, identified to the lowest possible taxonomic category, counted and weighed. Two-tailed t-tests or two-sample randomized tests were used to compare abundance and biomass between corresponding matched pairs of oiled and control sites for each taxonomic category at each MVD. Statistical comparisons of abundance and biomass were expanded to all sampled sites within each region, habitat type and MVD using Fisher's method (Sokal and Rohlf, 1982); statistical inferences were further extended to all possible locations within the universe of a given region, habitat type and MVD using Stouffer's method (Folks, 1984). Potential impact to the entire community at individual sites was examined using k-dominance curves, Shannon Wiener and Brillouin diversity

indices, and species richness and dominance Simpson indices. Some of the more noticeable trends are reported here.

Sites showed varying degrees of impact, probably due to such variables as the amount of oil, duration of exposure and amount of cleaning or bioremediation. Several sheltered rocky and coarse textured sites showed abnormal k-dominance curves, suggestive of stressed communities or communities in transition. Communities with abnormal k-dominance curves also had a higher dominance index, lower diversity index and lower evenness values. Examination of individual taxa revealed the following trends.

The most consistent differences in abundance and biomass between oiled and control sites in sheltered rocky habitat were observed for mussels, barnacles and limpets. The mussel, *Mytilus edulis*, had significantly higher abundance and biomass ($P < 0.05$) on control sites at MVD 2 and 3 in PWS and CIK during 1990. Comparisons at individual sites indicate that higher biomass and abundance resulted from unusually strong recruitment of mussels on control sites. Differences in *Mytilus* biomass and abundance were absent from PWS and limited to MVD 3 in CIK by 1991, possibly due to higher mortality of recruits on control sites, as indicated by the presence of larger individuals on control and oiled sites in both regions in 1991.

The barnacle *Chthamalus dalli* had higher abundance and biomass on oiled sites in PWS and CIK, particularly at MVD 2 and 3 in 1991 ($P < 0.05$). Abundance and biomass data indicate unusually high settlement on oiled sites, possibly due to greater amounts of free space created by cleaning and oil-related mortality of space competitors. Abundance

and biomass of the limpet *Tectura persona* was significantly higher on control sites at MVD 1 in PWS during both 1990 and 1991 ($P < 0.05$). Limpets can impact algae and barnacle recruitment by grazing down or bulldozing newly settled individuals.

Barnacle populations were much lower in coarse textured habitats; the dominant organisms included mussels, littorines (organisms living between high- and low-water marks) and oligochaetes (worms). The major trends are the following:

Mussel abundance and biomass showed similar trends in coarse textured habitats in PWS as occurred in sheltered rocky habitats. Significantly lower abundance and biomass on oiled sites at MVD 1 in 1990 ($p < 0.001$) apparently resulted from substantial mortality. Recruitment to MVD 1 on oiled sites in 1991 reduced the differences between oiled and control sites. Significantly higher mussel abundance on control sites at MVD 2 and 3 was apparently due to higher recruitment on control sites in both years. As in sheltered rocky habitats, elevated abundance on control sites was not accompanied by elevated biomass in 1991; these trends were apparently caused by higher growth rates on oiled sites and somewhat high loss rates of larger mussels on control sites. Mussels were present in low abundance on most coarse-textured matched pairs in CIK; statistical trends were moderate to weak or absent.

Littorine abundance was significantly higher on control sites at all MVD in PWS during 1990, but biomass was significantly higher on control sites only at MVD 1 ($p < 0.01$). Littorine populations increased by 3-5 times between 1990 and 1991 on three of the four oiled beaches, thus eliminating statistical differences at

MVD 1 and 3. Substantial increases in abundance of both *Littorina sitkana* and *L. scutulata* occurred, however, recruitment of *L. scutulata* was higher, particularly at MVD 3. Elevated abundance on control sites at MVD 2 in 1991 was due primarily to small individuals; biomass was significantly higher at MVD 2 on oiled sites ($p < 0.01$). Littorine populations were much lower in CIK and distinct trends were absent.

Oligochaete abundance was similar on control sites in both PWS and CIK, however, abundance was up to ten times higher on oiled sites in CIK than on control sites in 1990. There were substantial increases in abundance between 1990 and 1991 on oiled sites, thus suggesting continued impact as oligochaetes appear to benefit from oiling, particularly at MVD 3 in both regions.

Two of the most heavily stressed coarse-textured sites were Chugach Bay on the Kenai Peninsula and Snug Harbor in Prince William Sound. Both sites had an unusually high dominance by one or

two taxa. The total abundance of all animals in Chugach Bay at MVD 2 and 3 was 3752 - 4450 individuals per quadrat, over 90% of which were oligochaetes in 1990. The population at MVD 2 was still dominated by oligochaetes in 1991; the population at MVD 3 was dominated by oligochaetes and platyhelminths (flat-tented worms). The population in Snug Harbor was dominated by amphipods (crustaceans) and oligochaetes during 1990; oligochaetes continued to dominate the population at MVD 1 in 1991, but more diverse populations, similar to control sites, had begun to develop at MVD 2 and MVD 3. Thus, some areas were stressed or still undergoing recovery by the last sampling period in 1991.

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Coastal Habitat Injury Assessment: Intertidal Algal Communities

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The Coastal Habitat Injury Assessment program, part of the Natural Resource Damage Assessment plan, investigated the effects of the Exxon Valdez oil spill on the biota of the subtidal, intertidal, and supratidal habitats from Prince William Sound to the Alaskan Peninsula. Here we report the results of a study on the effects of the Exxon Valdez oil spill on the intertidal algal communities.

A random stratified experimental design was used to compare data from matched pairs of oiled and control sites in several different habitats in the three main areas of the spill: Prince William Sound, the Cook Inlet and Kenai Peninsula area (CIK), and the area including Kodiak Island and the Alaskan Peninsula (KAP). Each area was stratified by beach type: sheltered rocky, coarse textured, exposed rocky, and estuarine. Oiled sites were selected randomly from information made available from other agencies (Sundberg et al., 1993). Control sites were selected to match each oiled site using physical criteria such as beach slope, aspect, beach texture, and wave exposure. The overall statistical design was to compare data sets within site pairs and then to compare across all sites of one habitat from an area.

Sites were sampled during the field seasons in 1989 (late summer), 1990 (early summer, late summer) and 1991 (early summer, mid summer). Most sites had six randomly selected vertical transects with 3 quadrats (either 20 cm x 50 or 40 cm x 50 cm) located randomly within the

first, second, and third meter drop (MVD) from the high water mark. Percent cover of algal taxa was determined in 1991 from a 40- or 80- point grid in undisturbed (control) quadrats, quadrats cleared of all overstory algae, and quadrats scraped bare. Algae from scraped quadrats at all sampling times were preserved in formalin then sorted and weighed in the laboratory.

Because in all areas the most common alga, especially in the oil-affected upper intertidal, was the rockweed *Fucus gardneri* Silva, we performed several measurements on collected *Fucus* specimens to better ascertain potential damage to the population. In the field fertile receptacles were collected from randomly selected *Fucus* plants to determine the viability of released eggs. *Fucus* plants collected in cleared quadrats were measured for a variety of attributes including plant length, number and maturity of receptacles, and occurrence of damaged fronds, epiphytes, and regeneration.

Each data set was analyzed by site pair comparisons. Percent data were arcsine-square root transformed and tested for differences by Student's t-test. *Fucus* attributes were tested using either t-tests or a randomization test developed by WEST, Inc., Cheyenne, WY, from the two-sample randomization test algorithms by Manly (1990). Plate data were tested with a Fisher's exact test (Sokal and Rohlf, 1981). Overall tests of significance were tested with both a Fisher's combined test (Sokal and Rohlf, 1981) and by using Stouffer's consensus test

(Rice, 1990). In the discussion below all pairs of values are reported with the control value first, followed by the value from the oiled sites, and all p-values were calculated from the Fisher's combined test.

An overview of the data shows that in most habitats in the 3 areas *Fucus* showed significant differences between the control and oiled sites. However, each meter drop, each habitat, and each area had different patterns.

For the Prince William Sound area *Fucus* percent cover was significantly less at oiled sites in all habitats in 1991, more than 2 years after the spill. This pattern was found in the upper MVD in exposed rocky sites, in the upper 2 MVD's in both sheltered rocky and estuarine sites and at all 3 MVD's in coarse textured beaches. For example, in sheltered rocky sites *Fucus* covered 28.4% of the area in controls, but only 11.3% in the oiled sites ($p < 0.01$) in late summer 1991.

In the CIK area *Fucus* coverage generally showed more differences between oiled and control sites later in the summer in 1991. In sheltered rocky sites there was greater coverage of *Fucus* in controls (30.7% vs. 14.0%, $p < 0.01$) in the upper intertidal (1 MVD) but less coverage (16.4 vs. 37.2%, $p < 0.01$) 2 m lower (3 MVD). In coarse textured beaches *Fucus* coverage was not significantly different at any tide level in 1991. In estuarine sites there was greater coverage of *Fucus* in the upper 2 meters, but less coverage in the lower intertidal at control beaches.

The general pattern of more coverage by *Fucus* in control sites was repeated in the KAP area in the upper intertidal in sheltered rocky sites, and at all tide levels in coarse textured sites.

The lower coverage of *Fucus* in oiled

sheltered rocky and coarse textured habitats was complemented by the lower biomass of *Fucus* in the oiled sites in 1991. *Fucus* biomass averaged 1.2 kg m^{-2} in controls and only 0.36 kg m^{-2} ($p < 0.05$) at oiled sites in the upper intertidal sheltered rocky habitats in Prince William Sound. In coarse textured sites the differences were significant in the lower intertidal (3 MVD) zone (0.62 vs. 0.28 kg m^{-2} , $p < 0.05$).

Data from oiled sites indicate that the *Fucus* plants present in those sites were not as reproductive as those in control sites and suffered from a higher level of epiphyte infestation. The average *Fucus* plant at oiled sites at the second MVD was longer (2.5 vs. 4.4 cm , $p < 0.05$) due to the absence of smaller plants at the oiled sites. In the upper intertidal in oiled sites the *Fucus* plants had fewer receptacles (860 vs. 36 m^{-2} , $p < 0.01$), fewer receptacles per mature plant (16.5 vs. 9.2 , $p < 0.05$), and a lower reproductive index. However, egg viability was not significantly different between plants from control and oiled sites. *Fucus* from oiled areas had more adult plants with attached epiphytes (10% vs. 54% , $p < 0.01$) with a greater percentage of the area of each plant covered.

Because of the dominance of *Fucus* in the upper intertidal zones, the values for coverage of total algae and of perennials co-varies with the values for *Fucus*. The patterns for annuals and ephemerals varied with respect to the habitat and tide level. In Prince William Sound annuals and ephemerals were significantly greater in oiled sites with respect to both biomass and percent cover in rocky habitats in the upper intertidal in 1990 and early in 1991. By the end of the summer in 1991, there were no significant differences in these variables. In coarse tex-

tured sites there were initially no differences in annual and ephemeral coverage, except at the 3 MVD where coverage was higher in control sites. This pattern was retained through the end of the field season in 1991 in Prince William Sound. Estuaries in Prince William Sound had greater coverage of annuals and ephemerals in controls in early summer of 1991, but there were no differences by late summer.

The percent cover by ephemeral and annual algae in CIK in sheltered rocky and estuarine sites was similar to the trend in Prince William Sound but the differences between oiled and control sites persisted to the last visit in 1991. In coarse textured sites there were no differences in the cover of annuals and ephemerals in CIK between oiled and control sites, except for the last visit in 1991 at the 3 MVD where there was a slight trend for more annuals at oiled sites.

The KAP area had no significant differences between control and oiled sites with respect to coverage by annuals and ephemerals at any MVD at any habitat in 1991.

Other algae had varying responses to the oiling. Green algae that appeared to be adversely affected by the oiling and/or subsequent beach treatment (that is, those with percent cover values significantly lower in oiled sites) were bladed greens in all estuarine beaches, *Cladophora* in sheltered rocky sites in Prince William Sound, and *Acrosiphonia* in sheltered rocky sites in CIK. Coverage by filamentous browns was less in oiled coarse textured beaches in Prince William Sound and in oiled estuarine sites in Prince William Sound and CIK. Plants identified as *Myelophycus/Scytosiphon* had lower coverage in Prince William Sound

sheltered rocky oiled sites. Red algae that had lower coverage in oiled sites were *Halosaccion*, *Endocladia/Caulocanthus*, *Odonthalia*, *Palmaria*, and *Polysiphonia* in sheltered rocky sites in CIK, *Gloiopeltis* in sheltered rocky sites in Prince William Sound, *Cryptosiphonia* in exposed rocky habitats in Prince William Sound, and *Neorhodomela* in both exposed rocky sites in Prince William Sound and sheltered rocky sites in CIK.

A few algal taxa had greater coverage in the oiled sites than the controls. Members of the Gigartinaceae family had enhanced coverage in exposed rocky sites in Prince William Sound and in sheltered rocky sites in CIK and KAP. Brown algae that followed a similar pattern were *Myelophycus/Scytosiphon* in exposed rocky sites in Prince William Sound and filamentous browns in coarse textured beaches in CIK. In the red algae, *Gloiopeltis* had more coverage in oiled sheltered rocky beaches in CIK as did *Palmaria* in oiled exposed rocky beaches in Prince William Sound. *Cryptosiphonia* and *Odonthalia* had higher percent cover in oiled sheltered rocky habitats in KAP.

The Cook Inlet/Kenai area had the highest number of significant differences between oiled and control site pairs in the algal percent cover data. Most of these differences occurred at the sheltered rocky beaches. The Kodiak/Alaska Peninsula area had few significant differences, but many of the taxa that did show differences exhibited higher coverage in oiled sites. All areas had significant differences in the amount of uncolonized substrate. There was significantly more bare rock at oiled sites in all habitats at most tide levels in the three areas.

The perennial alga *Fucus* was the species most obviously affected by the Exxon

Valdez oil spill, especially in the upper intertidal. As of the end of the summer in 1991 coverage by *Fucus* was still significantly less in most oiled beaches. Our data indicate that recovery will be limited by the few mature plants in the area since the dispersal of *Fucus* zygotes is restricted to an effective diameter of 1 to 2 m (McConnaughey, 1985). The presence of a protective canopy is probably also a factor in enhancing the survival of *Fucus* germlings (van Tamelen and Stekoll, 1993). The length of time for the upper intertidal populations of *Fucus* to recover to the level of the control sites is unknown at present, but will be a function of the rate of dispersal from nearby *Fucus* beds.

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Damage and Recovery Rates of *Fucus* in Herring Bay, Knight Island

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The brown alga *Fucus gardneri* is the most abundant intertidal seaweed in Herring Bay, Knight Island, Prince William Sound, comprising up to 90% of the total algal biomass. The abundance of this plant and its simple and observable life cycle allowed detailed observations and experiments to be carried out regarding the consequences of the Exxon Valdez oil spill. Much of Herring Bay was heavily oiled, and a variety of clean-up technologies were performed throughout the bay. However, the southeast arm of the bay received little or no oil, allowing comparisons of oiled and unoiled areas within the same bay. Over three field seasons, 1990-92, we were able to quantify the damage done by the oil spill and associated clean-up to *Fucus* populations in Herring Bay. Using a variety of experimental techniques we also determined some of the factors influencing recovery of *Fucus* populations.

Five pairs of oiled and control sites were monitored, but for brevity and clarity we will report only the results from one representative pair of sites. The pair we will consider are two sheltered rocky sites. The oiled site was heavily oiled and probably received high pressure-hot water clean-up treatment. To assess damage to *Fucus* populations and assess recovery rates, we monitored 18 permanent plots (20x50 cm) at each oiled and control site. There were six randomly located plots in each of three tidal levels. Sampling consisted of estimating percent cover of all sessile organisms using a systematic point contact method. All

Fucus plants were then measured, and, if they were reproductive, the number of receptacles and stage of development were recorded.

In the upper intertidal, *Fucus* covered about 50% of the area at the control site, while at the oiled site *Fucus* cover was initially about 10%, increasing to about 20% in 1992. None of these differences were statistically significant due to the high variability in the data. In the mid intertidal, initially 80% of the area was occupied by *Fucus* at the control site, compared to 20% at the oiled site. By 1992 these significant differences had converged and were no longer statistically different. The percent cover of ephemeral algae was greater at the oiled site compared to the control. In the mid intertidal, about 20% of the area was initially covered by ephemeral algae, declining to almost none in 1991. Ephemeral algae were always scarce at the control site in the mid intertidal. In the low intertidal, ephemeral algae were initially more than twice as abundant at the oiled site compared to the control site, but this difference disappeared in 1991.

The differences in the percent cover of *Fucus* can be attributed to lower densities of large plants (>10 cm) and reproductive plants at the oiled site. In 1990 and 1991 large plants were almost nonexistent at the oiled site in the upper and mid intertidal. At the control site there were about 5 large plants per quadrat in the upper intertidal and 5-20 in the mid intertidal. In 1992, there were no significant differences between the oiled and

control sites in the number of large plants, but the upper intertidal recovery is not apparent. There were also few reproductive plants at the oiled site in the upper and mid intertidal. At the control site there were about 5 reproductive plants per quadrat at the same two tidal levels.

At the oiled site in the mid intertidal there were more germlings in late 1990 and early 1991. In 1991, there were more small plants and in 1992 there were more medium plants at the oiled site. This year class of plants probably settled the summer after the oil spill in 1989 and have grown into new size classes over the years, increasing the number of plants in subsequently larger size classes. It seems that recovery of *Fucus* in the mid intertidal is proceeding.

In order to assess the ability of *Fucus* to recolonize damaged sites the settlement rate of *Fucus* eggs was estimated. The relative number of *Fucus* eggs landing on oiled and control beaches was assessed by placing four Plexiglass plates designed to trap *Fucus* eggs at each of three tidal levels. By scoring the plates with a utility knife, grooves just wider than the diameter of a *Fucus* egg were made, *Fucus* eggs settle on the plates, concentrating in the grooves. These plates were left in the field for 24 hours after which the number of eggs on each plate was counted under a dissecting microscope. The 24 hour cycle was repeated for three consecutive days. These observations were performed three times in the summer of 1992. The distance to the nearest fertile *Fucus* plant was measured during all three sampling periods.

There were almost no eggs collected by the egg catcher plates at the oiled sites throughout the summer. At the control sites egg counts averaged up to 800 eggs

per plate per day. This difference in settlement rate can be explained in part by the low density of reproductive plants at oiled sites. The distance to the nearest fertile *Fucus* plant, which is inversely related to density, was about four times as great in oiled areas.

To assess *Fucus* recruitment and germling survival, unglazed ceramic plates were made and deployed in the field. These plates were made with grooves of three widths and two depths. Half of the plates were seeded in the lab with *Fucus* eggs, and the germlings were grown for a month before deployment. Four plates were placed at each of two tidal levels at three oiled and three control sites. After 2 months the number of germlings in each groove was recorded. Areas not in grooves on the plates were also monitored.

Seeded germlings only survived in the grooves of the ceramic tiles, and recruitment only occurred in the grooves. The widest, shallow grooves provided little protection for germlings from detrimental conditions. The narrowest grooves were slightly smaller than *Fucus* eggs and *Fucus* did not recruit well to these grooves. Medium-width grooves and deep, wide grooves provided good habitats for young *Fucus* plants.

Experiments performed in 1991 using petri dishes instead of ceramic plates showed that germlings usually did not survive more than 1 month on flat surfaces. To determine possible mechanisms of this high mortality, the survival, estimated percent cover, of these germlings was compared to the desiccation rate at the site of the petri dishes. Also, four of these dishes were experimentally exposed to whiplash from adult plants. Desiccation rates were greater at oiled sites, and desiccation was negatively cor-

related with percent cover of seeded germlings in petri dishes. Where desiccation was greater, *Fucus* germlings were less abundant due to higher mortality. Adult *Fucus* canopy can lower desiccation stress, potentially enhancing germling survival. However, when germlings on plates were placed under *Fucus* canopy without herbivores and in constant contact with the ocean, very few germlings survived after 2 weeks. Whiplash from the adult plants removed the germlings from the substrate. By providing a refuge from desiccation and whiplash from adult plants, cracks and crevices in the substrate seem to greatly enhance germling survival.

The growth rates of established *Fucus* plants were determined by marking randomly chosen individual plants in each of three size categories at three tidal levels. The size categories were 2.0-4.5 cm, 5.0-9.5 cm, and >10 cm. These plants were measured to the nearest 0.5 cm in spring 1991, fall 1991, and summer 1992. New plants were tagged when mortality or loss of tags occurred.

In the upper intertidal in oiled areas *Fucus* plants in all size classes grew about twice as fast as plants in control sites over a period of 1 year. In the mid intertidal only large plants grew faster in the oiled areas. This indicates that once *Fucus* plants become established in oiled areas recovery may proceed rapidly.

Results from our experiments and observations showed that large *Fucus* plants in the upper and mid intertidal showed lower densities in oiled sites compared to control sites. Fewer large plants created less cover of *Fucus* and more open space for ephemeral algae to colo-

nize. Ephemeral algae showed greater abundances up to 2 years after the spill at oiled sites. Since reproductive *Fucus* plants are usually at least 10 cm in length, fewer large plants meant lower densities of reproductive *Fucus* at oiled sites. Since *Fucus* eggs have limited dispersal, the lack of reproductive plants has led to observed lower *Fucus* egg settlement at oiled sites. Heterogeneity in settlement substrata was required at both oiled and control sites for recruitment of settled eggs into the *Fucus* populations. At oiled sites, cracks and crevices reduce the effects of heat and desiccation, while at control sites, whiplash from adult plants was ameliorated by cracks. Thus, surface heterogeneity is almost essential for *Fucus* to recruit into damaged and unoled areas. New recruits in upper zones of oiled areas were confined to relatively deep cracks in the rock surface, and casual observations revealed that almost all *Fucus* plants are found to be attached in cracks.

Due to low settlement rates and severe environmental conditions recruitment of *Fucus* into areas severely damaged by the oil spill and associated clean-up efforts, particularly the upper intertidal, has been minimal. In areas with less harsh conditions, mid and low intertidal zones, *Fucus* has recruited abundantly. Once recruited to a damaged area *Fucus* plants grow faster due to reduced intraspecific competition since few plants remain at oiled sites, especially if those sites were heavily cleaned. Thus, where recruitment has occurred, recovery of *Fucus* populations is proceeding rapidly. However, where *Fucus* recruitment is low, recovery has been slow.



Meiofaunal Recolonization Experiment with Oiled Sediments; The Harpacticoid Copepod Assemblage

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To gain insight into the effects of the *Exxon Valdez* oil spill on the meiobenthos (small organisms living on the ocean floor) of Prince William Sound, Prudhoe Bay crude oil was used in a colonization study initiated in 1990. Sediment, collected near Juneau, Alaska, was repeatedly frozen and thawed, washed with freshwater and sieved through 2 and 0.417 mm screens to kill and remove meiofauna. Prudhoe Bay crude oil was added and mixed into this azoic sediment to reach concentrations of 0.5 and 1.7% crude oil. The resulting mixture was added to replicated colonization trays (13 x 28 x 33 cm). Non-oiled azoic sediments were added to additional trays. Triplicate trays of all treatments were placed flush with the sediment surface on beach transects in a randomized block design near mean low water (-0.6 m) in Herring Bay (a cove that was heavily oiled from the *Exxon Valdez* spill), Prince William Sound. Trays were sampled by coring on days 0, 1, 2 and 29. Cores were also collected from nearby undisturbed ambient sediments on each collection date, including Day 0, to quantify the colonizing source pool. Here, we report on an analysis of the harpacticoid copepod species from these collections.

Collections of meiofauna taken from azoic sediment prior to placement in the field (Day 0) suggested that although meiofauna were killed, decomposition was incomplete. Copepods, with chiti-

nous cuticles, were especially resistant. Residual copepods in our colonization samples were identified by observation of the condition of the setae of various appendages, with particular attention given to the caudal setae; a large number of broken setae indicated to us that these individuals were dead when collected. Most copepods with broken caudal setae showed signs of decomposition including ruptured cuticles, partially decomposed flesh and a dense detrital coating over the body. All copepods were recorded as "dead" or "alive" at the time of collection based on this observation. Generally, the number of "dead" copepods was roughly equal in all experimental treatments (high, low and control) on all collection dates. An ANOVA conducted on data from Day 29 did not identify an oiled treatment effect suggesting that dead copepods were residuals in all treatments including the controls, and that the density of "dead" copepods was not related to oil-induced mortality. Dead copepods were rare in ambient sediments.

Harpacticoids were diverse with > 40 species encountered. The density of copepods increased in ambient sediments throughout the collection period. Highest densities of copepods in colonization trays were also observed in Day 29 collections. Species analysis indicated that sediments and colonization trays were occupied primarily by copepods associ-

ated with surrounding eel-grass and algal-mat habitats. Most species displayed strongly prehensile first legs, belonging to families associated with a phytal lifestyle. Day 1 and 2 collections demonstrated that colonization was generally rapid (mean densities in trays were similar to those from ambient sediment collections). No single species or succession of species accounted for the rapid colonization; instead colonists belonged to a large number of taxa each found in rather low densities.

A two-way ANOVA was conducted as a randomized block (tray replication was blocked) design. ANOVA treatments were collection day (1, 2 or 29) and oil treatment (control, 0% oil, low, 0.5% oil or high, 1.7% oil). Tests were performed on the total living copepods, unidentified copepodites, and the five most abundant species, two of which were in the Family Diosaccidae, and one each in the Families Canthocamptidae, Laophonitidae and Ectinosomatidae. Collection day effects were significant in all taxa tested reflecting the increase in abundances from Days 1-29 (density in ambient collections increased for the same time). A treatment effect was observed in total copepods, and species designated *Canthocamptus* sp. and Ectinosomatidae sp. 1. For total copepods and *Canthocamptus* sp., Tukey's Studentized Range Test indicated that densities in control and low-oiled sediments did not differ, but that densities in high-oiled sediments were significantly lower than other treatments. For Ectinosomatidae sp. 1, means were significantly different in all experimental treatments, but densities in control sediments were significantly higher than in low-oiled treatments and, in turn, densities in low-oiled were significantly higher

than in high-oiled treatments. Our data suggest that differences in colonization rate (either immigration or emigration), rather than oil-induced mortality, were the cause of this difference.

Detrended correspondence analysis was conducted to determine if an oiling effect influenced the copepod assemblage. Results indicate that two axes comprised > 70% of the variance. Axis one contained relatively high levels of variation, with values approaching two standard deviations overall (four indicates complete faunal turnover). All natural samples separated from experimentals on axis one and clustered near to each other indicating that experimental trays could be distinguished from the surrounding sediments in species composition on all collections. Other collections were separated on axes 2 with only 1 standard deviation of faunal turnover. Day 1 and 2 low- and high-oiled treatments clustered together as did control, low- and high-oiled sediments from Day 29. Control Day 1 collections were intermediate. Results therefore indicate that an oiling effect was present on Days 1 and 2, but no difference between control and oiled sediments was apparent by Day 29. Species diversity and evenness were similar in all treatments, suggesting that there were no oil-related effects on the number of species colonizing the experimental trays.

Knowledge of the ability of organisms to recolonize an area following an oil spill is vital to understanding the impact of such spills. The response of benthic animals to hydrocarbons varies (Fleeger and Chandler, 1983; Coull and Palmer, 1984; Coull and Chandler, 1992). A combination of effects on mortality, reproduction and migration may all contribute to observed changes in density.

Our data suggest that copepods have the ability to recolonize azoic sediments over small spatial scales (many cm²) quickly following the addition of hydrocarbons (see also Alongi et al., 1983; Decker and Fleeger, 1984). Meiofauna are known to be active colonizers through the water column, especially in muddy sediments (Chandler and Fleeger, 1983; Palmer, 1988), and many individuals colonized our experimental trays after one day, even in highly-oil sediments.

However, hydrocarbons, especially at high concentrations, altered colonizing ability. The abundance of two individual species and total copepods was significantly depressed at high-oil dosages. The effect was short-lived however, as a influence of high-oil dosages on community structure was apparent on Days 1 and 2 of the experiment, but not identifiable after 29 days.

We could identify no strong evidence for oil-induced mortality in our colonization trays; the number of "dead" copepods was not different in oiled- compared to non-oiled trays on any collection. Copepods may have avoided the oil-enriched trays or they may have emigrated from the trays at faster rates. Recent data suggest that copepods that have entered the water column have the ability to select specific locations to colonize (Fegley, 1988; Sun and Fleeger unpublished data), but only under low flow conditions.

Emigration rates could also have been affected by hydrocarbons. Copepods under high-oil conditions may display different behaviors; they may have actively emigrated by swimming into the water column or passively emigrated by

allowing themselves to be carried away by tidal currents.

In summary, abundance data collected after an oil spill cannot alone determine if migration or birth and death processes are responsible for changes in density. Our data suggest that meiofauna are rapid colonizers, but at high doses of oil, colonization rates are significantly reduced. Better information on the source pool of immigrating copepods is needed to help interpret density changes in a given locale after a spill of the magnitude of the *Exxon Valdez*.

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Influence of the Exxon Valdez Oil Spill on Intertidal Algae: Tests of the Effect of Residual Oil on Algal Colonization

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Following the Exxon Valdez oil spill in 1989, we began to study the effects of stranded oil on algal colonization in rocky intertidal communities in Prince William Sound. This knowledge is critical because (1) algae are extremely important structurally and functionally to intertidal communities; (2) there appears to be a predictable succession of organisms following a spill, and enhancement of the early phases could reduce the time to recovery; (3) there is evidence that oil inhibits algal growth (Straughan, 1971); and (4) there is evidence that cleanup measures on rocky shores might even delay recovery, if their biotic effects are as harmful as those of the oil (Thomas, 1978; Foster et al., 1990). As part of recovery from a spill, succession on oiled substrates can differ from natural succession, due to residual toxicity of the oil, large scale mortality that reduces available propagules, and loss of herbivores (Southward and Southward, 1978). We were particularly interested in whether normal mechanisms of colonization were affected by the stranded Exxon Valdez oil, so we examined the initial stages of a succession, the settlement, growth, and early survivorship of algae.

Rocks or tiles were used as substrates for colonization. In 1989, oiled rocks of a similar size were collected from beaches on Knight and Eleanor Islands. In 1990, oiled rocks were collected from the western arm of Herring Bay. In addition,

clean rocks were collected from the supratidal zone in southeast Herring Bay, coated with fresh Prudhoe Bay oil, and allowed to weather for a month until the oil was a tarred consistency. Ceramic tiles were similarly prepared with Prudhoe Bay crude oil.

To account for inherent patchiness in algal colonization, we paired oiled and unoled substrates in each experiment. In addition, the Herring Bay experiments were set up on carefully matched oiled and unoled beaches. The oiled rocks had one half of the upper surface area cleaned with methylene chloride and oiled tiles were paired with unoled tiles. All substrates were placed in the rocky intertidal. Prepared rocks from Knight and Eleanor Islands were placed on Gull Island, an unoled site southeast of the grounding site, and retrieved in 1990. The tiles and the Herring Bay rocks were placed on 6 beaches (3 oiled, 3 unoled) in Herring Bay and retrieved in 1991. Also in 1991, new oiled and unoled tiles (unglazed) were placed, ridged side up, on the 6 Herring Bay beaches and a new treatment was added. Half the tile pairs were in cages designed to exclude grazers and half were unprotected. Settlement on these tiles was measured 1992.

In most cases, the oiled sides of rocks and oiled tiles had been colonized by significantly fewer algae than the clean surfaces. On average, algal coverage on oiled sides was less than one third the

coverage on the clean sides. In 1990, for example, the half-cleaned rocks on Gull Island had approximately 70% less algal cover on the oiled sides than the cleaned sides (paired t-test; $p=0.001$). In Herring Bay in 1991, many of the rocks had been lost, so only the tile data are presented here. For the six beaches, paired t-tests revealed consistently lower algal cover and fewer *Fucus* germlings on oiled tiles than on unoiled tiles, but at p levels that were generally 0.1 to 0.2. These results prompted the caging experiment in 1991. The 1992 results for the caged tiles also showed a reduction in algal cover, with significant differences (paired t-test; $p<0.05$) detected at several beaches. For the other beaches, p ranged from 0.06 to 0.17 for *Fucus* germlings. Testing the combined probabilities from the separate tests (Sokal and Rohlf, 1981) revealed that oil significantly reduced the settlement of *Fucus* germlings ($p<0.001$; χ^2 test with 12 d.f.). The effect of oil on settlement on the exposed tiles was not as consistent.

Excluding grazers enhanced the ability to detect statistically significant effects of the oil in Herring Bay. Grazers were not controlled on Gull Island or the 1990 studies in Herring Bay. Grazers feeding preferentially on the oiled sides of rocks and tiles could have accounted for our results. However, we hypothesize that given the susceptibility of grazers to stranded oil (Peltier et al., 1976; Crapp, 1971), the grazers would probably feed preferentially on the unoiled substrates, which would mean we may have underestimated the effect of the oil. Indeed, in our 1991 study, we saw significantly more limpets and littorinids on unoiled tiles than oiled tiles ($p<0.001$; combined probabilities from separate paired t-tests, χ^2 test with 48 d.f.; Sokal

and Rohlf, 1981).

The effect of oil on algal colonization was very pronounced and consistent across a variety of locations in Prince William Sound and from year to year as well as across methods. The two-thirds reduction in percent cover indicates how sensitive this stage of the natural recovery process was to even residual levels of oil. The paired design helped reduce variability due to differences in surface texture of rocks and tiles, and heterogeneity of microenvironments on Gull Island and in Herring Bay. The reduced colonization on oiled substrates in our experiments is similar to that previously described (Nelson, 1981; Notini, 1978; Southward and Southward, 1978), suggesting that inhibition of initial recovery reported from other spills was operating at our sites in Prince William Sound.

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Variability of Exxon Valdez Hydrocarbon Concentrations in Mussel Bed Sediments

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Concern for mussel beds contaminated by the *Exxon Valdez* oil spill increased rapidly in 1991 when poor recovery in several predator species was thought to be linked to oiled mussels. A pilot survey confirmed the persistence of *Exxon Valdez* oil at relatively high concentrations in mussels and underlying sediments in several beds (Babcock et al., these proceedings). The uneven distribution of oil within beds observed during the survey prompted us to sample several beds intensively in 1992 to determine the within-bed variation in sediment hydrocarbon concentrations. In this paper we document concentrations and distribution of hydrocarbons within one bed and examine the effects of several variables.

The study site, on the northern tip of Chenega Island, is fairly typical of highly oiled beds. It is small (approximately 50 m²) and on a low angle beach (4.4% slope) protected from intense wave action by bedrock headlands. Tidal range occupied by the bed is approximately 1.43 m to 1.77 m above mean lower low water. Mussel densities average 1900 animals per m² on sediments ranging from small pea gravel to fine silt.

At each of the 15 subsites within this bed, mussel density was determined and sediments entrained in the mussel byssal mat (threadlike mass holding mussels together), and sediments under this mat to a depth of 2 cm were sampled for hydrocarbons. All sediment samples

were extracted and analyzed by UV spectrophotometry at the Auke Bay Laboratory. This method, adapted from Krahn (1991), approximates total oil concentrations based on the concentrations of two and three ring aromatic compounds that fluoresce at the phenanthrene wavelengths (260/380 nm). Although analytical results are not strictly quantitative and can not be compared with results produced by GC-MS analysis, they are extremely useful in comparing large numbers of samples.

Oil concentrations in sediments underlying the mussel bed were highly variable, ranging from 20 µg/g wet weight to 40,498 µg/g with a mean of 13,662 µg/g (S.D.=13,326 µg/g). The effects of position on the beach and depth were tested with a three way ANOVA (beach position has two components: y axis represents tidal height, x axis is at right angles to y and represents distance from the bed's central axis). Oil concentrations were significantly greater ($P < .05$) at the two lower tidal heights, 1.50 m and 1.59 m; mean oil concentrations were 25,778 µg/g and 26,403 µg/g respectively. At the upper tidal height, 1.73 m, mean oil concentration was 1515 µg/g. A two way split plot analysis, treating the two sediment depths sampled at a subsite as two treatments applied to one plot, determined that at each subsite oil concentrations in byssal mat sediments (mean=9383 µg/g) were significantly lower than in sediments collected below

the mat (mean=17940 μ g/g).

Mussel densities ranged from 15 to 244 animals per sample quadrat (625 cm²) with a mean of 120 mussels per quadrat (1900 per m²) (S.D.=72). The ANOVA found no significant relationship between mussel density and oil concentrations in either byssal or underlying sediments. This finding is not at odds with the idea that mussels insulate underlying sediment hydrocarbons from tidal flushing. The stability provided by even a sparse mussel cover may be sufficient to allow high levels of hydrocarbons to persist in sediments.

Sediments appear to be a complex reservoir of oil that continues to contaminate mussels. The UV analyses of mussel bed sediments have confirmed visual observations that oil in the study bed is unevenly distributed and that concentrations are higher at lower tidal heights and below the byssal mat. Given that the physical distances are minimal between the lower and upper tidal heights and between sediment depths,

the apparent effects of both depth and tidal height on oil concentrations may be related to a third factor, sediment grain size. Further analysis will examine these relationships. The variability of mussel hydrocarbon concentrations within a bed and the relationship to contamination in directly underlying sediments will be clarified when GC-MS mussel analyses become available.

Given the current level of contamination three years after the *Exxon Valdez* spill, the decline of hydrocarbon concentrations in mussel beds to pre-spill levels will be a prolonged process. In monitoring that recovery, sampling must account for high within-bed variability.

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Oiled Mussel Beds Two and Three Years after the *Exxon Valdez* Oil Spill

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In 1991, two years after the *Exxon Valdez* oil spill, scientists observed crude oil associated with some mussel beds that still smelled of fresh aromatic hydrocarbons. Coincidentally, biologists observed continued reproductive failure among harlequin ducks and oystercatchers in the spill area, and possible reduced survival among young sea otters and river otters. All of these higher order consumers are dependent on mussels (*Mytilus trossulus*) for a large portion of their diets.

We conducted surveys to determine the geographic distribution oiled mussel beds and the concentrations of oil in the beds inside Prince William Sound and along the northwestern shoreline of the Gulf of Alaska. Thirteen mussel beds with evidence of oil present were located in Prince William Sound in 1991, and samples of mussels and underlying sediments were taken from each for analysis of aromatic hydrocarbon content.

In 1992, we resampled most of the 1991 sites and located and sampled 46 additional oiled mussel beds in Prince William Sound. At some sites we sampled more than one mussel bed. Cooperating in the survey and sampling were the Alaska Department of Fish & Game, Alaska Department of Environmental Conservation and the U. S. Fish & Wildlife Service.

Sampled oiled mussel beds within Prince William Sound encompassed primarily the Knight Island group but were actually bound by Green Island on the

eastern side, Naked Island on the north, Applegate Island on the northwest, and the Fox Farm site on Elrington Island on the south.

Five control sites were also sampled (Barnes Cove on Knight Island, Olsen Bay, Crab Bay on Evans Island, and West Bay on Bligh Island); extensive histories of petroleum hydrocarbon concentrations in mussels and sediments exist for all these control sites.

Forty mussel beds were evaluated on the Kenai Peninsula, Kodiak Island, and in Katmai National Park and Preserve; oil was observed and mussels and sediments sampled at 13 of these sites. Sampled sites ranged from Tonsina Bay on the Kenai Peninsula, to Cape Nukshak in the Katmai National Park and Preserve.

Criteria for sampling mussels and underlying sediments were the appearance and smell of crude oil in the sediments immediately underlying a moderate to dense mussel bed. Triplicate sediment samples (each consisting of a composite of 8-10 subsamples) were taken along a 15-50 m transect line laid through the densest part of the mussel bed, generally 0-2 cm immediately underneath the mussels. Triplicate, pooled mussel samples (20 mussels each) were collected from the same areas as the sediment samples.

The 1991 mussel and sediment samples were analyzed using gas chromatography/mass spectroscopy (GC/MS) and units are reported as µg/g total

aromatic hydrocarbons. Sediment samples collected in 1992 were analyzed using a UV fluorescence screening procedure adapted from Krahn et al. (1991). Excitation/emission spectra of the extracts were read at the phenanthrene wavelength (260/380 nm), and values reported are $\mu\text{g/g}$ wet weight total oil equivalents (OE). This procedure does not measure individual analytes within a sample, but does approximate total oil concentration and allows comparison of relative oil concentrations between samples. The UV screening permits the rapid analyses of more samples than by the costly GC/MS procedure.

In 1991 samples, the highest total aromatic hydrocarbon concentrations in mussels were collected from Foul Bay on the Western mainland (mean = 10.31 ± 2.9 standard error $\mu\text{g/g}$ dry weight), Bay of Isles on Knight Island (mean = 6.0 ± 1.1) and Northeastern Latouche Island (mean = 3.8 ± 1.3). Highest concentrations of total aromatic hydrocarbons were shown in underlying sediments collected from the north end of Chenega Island (mean = 427.2 ± 29.4 $\mu\text{g/g}$ wet weight), a tombolo on Eleanor Island (mean = 36.1 ± 24.3) and Bay of Isles (mean = 28.7 ± 9.4).

In sediment samples collected in 1992, 19 mussel beds in Prince William Sound had concentration in excess of 10,000 $\mu\text{g/g}$ wet weight oil equivalents. The UV analyses of sediments collected in 1992 indicated the highest petroleum hydrocarbon concentrations from an islet in Foul Bay (mean = $62,258 \pm 1,558$ $\mu\text{g/g}$), a small tombolo on an islet in Herring Bay (mean = $39,394 \pm 8,655$), and the east-

ern shore of Applegate Island (mean = $30,394 \pm 1,081$). Mean concentrations of sediments from 3 control sites were less than <3 $\mu\text{g/g}$ OE.

Sediments from mussel beds in the Gulf of Alaska show highest levels from Port Dick (mean = $9,122 \pm 2,312$ OE), Tonsina Bay ($8,250 \pm 2,793$), and Windy Bay ($4,645 \pm 1,169$)—all along the Kenai Peninsula.

No mussel samples collected in 1992 have been analyzed.

The limited data available to compare levels between 1991 and 1992 suggest that petroleum hydrocarbon concentrations in sediments beneath the layer created by moderately to densely packed mussels are relatively unchanged, indicating that natural processes are only slowly cleansing these beds. This means that the mussels will probably experience chronic oil exposure for years.

We have documented 31 mussel beds within Prince William Sound and 9 along the Kenai Peninsula and Alaska Peninsula showing sediment petroleum hydrocarbon levels in excess of 1700 $\mu\text{g/g}$ wet weight oil equivalents. The potential continued contamination of overlying mussels which form an important food source for higher consumers needs to be closely monitored for natural recovery or for possible use of manipulative restoration measures.

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Determination of Petroleum-Derived Hydrocarbons in Seawater Following the Exxon Valdez Oil Spill II: Analysis of Caged Mussels

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We deployed bay mussels (*Mytilus trossulus*) that were initially free of hydrocarbons in nearshore waters along the path of oil spilled by the T/V *Exxon Valdez* to determine the persistence and the biological availability of petroleum-derived hydrocarbons to living marine resources. Mussels filter substantial volumes of seawater, and may therefore accumulate petroleum hydrocarbons integrated over the transplant period.

Petroleum hydrocarbon-free mussels were collected from Admiralty island in southeastern Alaska, and were transplanted to 12 locations inside Prince William Sound and to 18 locations outside the Sound for 2 to 6 weeks at depths of 1, 5, and 25 meters at each location. In Prince William Sound, four successive transplants were conducted in both 1989 following the spill and in 1990; two transplants were conducted during 1991.

Three successive transplants to sites along the Kenai Peninsula, Alaska Peninsula, and Kodiak Island occurred in 1989 and 1990. Mussels were retrieved at the end of each transplant period and stored frozen at -20°C for petroleum hydrocarbon analysis.

Transplanted and control mussels were analyzed using single ion mode gas chromatography-mass spectrometry (GCMS/SIM) for the most abundant 2- to 5-ring polynuclear aromatic hydrocarbons (PAH's) in the spilled oil, and using gas chromatography-flame ionization detection for alkane hydrocarbons including pristane, phytane, and the nor-

mal alkanes of 10 to 30 carbon atoms.

Results indicate that mussels transplanted along the trajectory of the oil spill accumulated particulate oil at concentrations that decreased with depth, elapsed time after the spill, and distance from heavily oiled beaches. The highest concentration of total PAH's in the transplanted mussels was 5.70 ± 0.358 standard error $\mu\text{g/g}$ wet tissue weight at Herring Bay, 1 meter depth, 1 to 2 months following the spill; at the 5 and 25 meter depths, concentrations were $3.17 \mu\text{g/g}$ and $0.372 \pm 0.139 \mu\text{g/g}$, respectively. Concentrations of PAH nearly as high at the respective depths were also found at north Smith Island and at Snug Harbor.

The mussel transplant sites at each of these three locations were within 500 m of beaches that had been heavily oiled by the spill. The relative concentrations of PAH and of alkane analytes detected are generally consistent with those of *Exxon Valdez* oil (EVO), indicating up to $281 \mu\text{g}$ EVO/g wet tissue weight.

Lower but detectable PAH concentrations were observed at most other transplant locations within PWS, with relative concentrations of PAH and of alkane analytes that are generally consistent with those of EVO. However, the lowest PAH concentrations were found at the control site, Olsen Bay, where total PAH concentrations generally ranged from $.010$ to $.050 \mu\text{g/g}$, with relative concentrations that are not consistent with those of EVO.

Concentrations of PAH's consistent

with EVO inside Prince William Sound declined substantially at all locations by late summer 1989. Total PAH concentrations of up to 1.47 $\mu\text{g/g}$ were observed at Herring Bay, and were at least an order of magnitude lower at north Smith Island or at Snug Harbor. Still lower concentrations of total PAH's consistent with EVO were detected at most of the remaining locations inside Prince William Sound, although not at Olsen Bay.

Low concentrations of PAH's were sporadically detected at locations where the transplant sites were adjacent to heavily oiled beaches in 1990 and in 1991. Total PAH concentrations of about 0.26 $\mu\text{g/g}$ were detected at 1 m depth at Herring Bay and at Snug Harbor in 1990, while the highest PAH concentrations detected in 1991 were near detection limits.

Petroleum hydrocarbons were detected only sporadically in mussels deployed at locations outside Prince William Sound in 1989, and were generally below detection limits in mussels de-

ployed during 1990 and 1991. This may have been due in part to poorer survival of the mussels transplanted to locations outside Prince William Sound, resulting from longer transport times.

The accumulation of petroleum hydrocarbons by the transplanted mussels in 1989 indicates that particulate petroleum hydrocarbons were generally available to subsurface marine fauna the summer following the spill. These results are consistent with, and support, results of a companion study where direct chemical analyses of subsurface seawater for petroleum hydrocarbon were performed on samples collected 1 to 6 weeks following the spill: both these studies found the highest concentrations of PAH's attributable to EVO at the 1 m depths of sites adjacent to heavily oiled beaches.

However, comparison of the results of these two studies indicates that the caged mussels may accumulate petroleum hydrocarbons from much lower seawater concentrations than may be detected by direct chemical analysis.



Estimation of the Exposure Concentration of the Seawater Soluble Fraction of Crude Oil from Mussel Tissue Concentrations

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Mussels have been used for many years to monitor petroleum hydrocarbon pollution in the marine environment (Farrington, 1980). Mussels appear to be ideal living monitors as these filter-feeders process large amounts of water, rapidly accumulate oil to high concentrations without apparent toxicity and rapidly purify the accumulated body burden without appreciable metabolism of the oil hydrocarbons when placed in clean seawater. For mussels to be useful in monitoring environmental contamination, the tissue concentrations must accurately reflect the magnitude of contaminants in the water. Despite a large number of studies devoted to quantitating oil hydrocarbon concentrations in mussels as a measure of water pollution, very few laboratory or field studies have attempted to correlate oil concentration in water with tissue concentrations.

Recently, however, a careful study has documented a relationship between water and tissue concentrations (Mason, 1988, 1 & 2). The extensive laboratory data from this study using black mussels and the water soluble fraction of Quatar crude oil were carefully analyzed using a well studied mathematical uptake model which relates the oil in water concentration and the uptake and purification rate constants with tissue concentrations over time. Since the major objective for monitoring mussels is to determine the pollution concentrations in the water, there is a need for a simple method to estimate the exposure concentrations based on

tissue concentrations.

The work described in the present paper extends Mason's methodology to provide a rapid simple method to estimate exposure concentrations of oil in seawater from previously determined oil concentrations in mussel tissue. The uptake period was shortened to 12 hours using small incubation volumes and frequent replacements of the water soluble fraction of *Exxon Valdez* crude oil to better provide a constant exposure concentration. In addition, the uptake data was analyzed using an equivalent but alternative mathematical uptake model to derive the uptake constants and provide a direct method for estimating exposure concentrations from predetermined concentrations of oil in the mussel tissue (Spacie, 1983).

Mussels (*Mytilus edulis*) were obtained from Hood Canal, WA, and maintained in 2.5% artificial seawater (Instant Ocean) at 11°C. The water soluble fraction (WSF) of *Exxon Valdez* crude oil was prepared by shaking 25 ml of oil with one liter of 2.5% artificial seawater for five minutes at 11°C. The phases were allowed to separate for 21-22 hours in the cold and the seawater phase drained from the separatory funnel into a glass stoppered bottle. The WSF was immediately diluted to 0.1%, 1%, and 10% for the mussel experiments.

Fluorescence was used to quantify oil hydrocarbons in unextracted or hexane-extracted water. The samples were excited at 280 nm and the emission read at

374 nm with no barrier filter in a Hitache MPF-2A analytical fluorescence spectrophotometer (Mason, 1987). Fluorescence was recorded in centimeters of pen deflection. Hydrocarbon concentrations in unextracted water were determined directly by fluorescence for the time course studies on the effect of aeration of the WSF in the absence of mussels. Seawater samples taken from the flasks containing mussels were centrifuged at 1000 xg for 10 minutes at 11°C to remove debris prior to hexane extraction. The volume of water extracted with 5 ml hexane was 40 ml for the 0.1% WSF exposure; 5 ml for the 1% WSF exposure; and 0.5 ml for the 10% WSF exposure.

Hexane extraction was necessary to eliminate nonspecific fluorescence produced by the mussels and to concentrate the hydrocarbons sufficiently to be readable in the fluorimeter. The hydrocarbon concentrations in the hexane fraction of the water taken from the flasks containing mussels were determined by fluorescence from a standard curve of *Exxon Valdez* crude oil diluted in hexane. The concentration of hydrocarbons in undiluted crude oil was determined to be 600 mg/ml by gravimetric analysis. In addition to the quantitation of hydrocarbons in the WSF by fluorescence, the concentration of aromatic hydrocarbons (AHC) in the C12-C24 range in the WSF preparations was determined by gas chromatography (GC) (Analytical Resources, Inc., Seattle, WA).

Time course experiments with mussels were performed by exposing 10 mussels/flask (approximately 5 g/mussel) to one liter volumes of 0.1%, 1%, and 10% WSF at 11°C. The flasks were gently aerated. The one liter volumes of WSF were completely replaced with fresh WSF every two hours for a total time course of

12 hours. Hydrocarbon concentrations in the water at the end of each two hour incubation were then determined. Since mussel tissue has not been analyzed at this point in time, uptake by the mussels was determined from the amount of hydrocarbons which disappeared from the water over time assuming the disappearance from the water equals uptake by the mussels. Mussels retrieved at two hour intervals were wrapped in aluminum foil and frozen at -20°C for later analysis.

Fluorescence of 2-fold dilutions of the water soluble fraction (WSF) of *Exxon Valdez* crude oil in 2.5% artificial seawater was directly proportional over a 100-fold range (1:8 - 1:1024 dilution). The regression was statistically significant ($r = 0.996$, $p < 0.01$). This plot suggests oil hydrocarbons soluble in seawater can be reliably detected at very low concentrations. To determine the concentration of water soluble crude oil in the seawater and in the mussel tissue, a fluorescence standard curve was obtained using 10-fold dilutions of *Exxon Valdez* oil in hexane. The logarithm of fluorescence was linearly related to the logarithm of the hydrocarbon concentration over a 100-fold range (0.001 - 0.1 µg/ml). The regression was highly significant ($r = 0.945$, $p < 0.01$). The concentration of oil in the WSF preparations used for the mussel uptake experiments determined from this standard curve were 4.27 µg/ml for the experiments using 0.1% and 10% WSF and 2.35 µg/ml for the experiments using 1% WSF.

The stability of the fluorescent hydrocarbons in the 10% WSF preparation under conditions of aeration was determined by fluorescence. Aeration of the 10% WSF in the absence of mussel resulted in a 6.4%/day loss of fluorescence ($r = 0.965$, $p < 0.01$) with no statistically

significant loss in the unaerated control over a six day period. In contrast, ten mussels in one liter aerated volumes of 0.1%, 1%, and 10% WSF removed approximately 90% of the fluorescent hydrocarbons in two hours at 11°C. The percentage loss of fluorescent hydrocarbons from the water was similar for each of the six consecutive two hour incubations which constituted an individual 12 hour time course. These results suggest that the uptake rates were rapid and uniform for 0.1%, 1%, and 10% WSF exposures throughout the 12 hour uptake period and saturation was not evident.

The petroleum hydrocarbon concentrations remaining in the hexane extracted water were determined from the oil in hexane standard curve using fluorescence. By assuming the loss of fluorescent hydrocarbons from the water equaled the accumulation of oil hydrocarbons in the mussel tissue, uptake values for mussels could be calculated. The tissue mass was taken to be 50 grams. For the three 12 hour time courses with complete renewal of the WSF at two hour intervals, the accumulated 12 hour uptake values for 0.1%, 1%, and 10% WSF were 0.442, 2.80, and 44.7 $\mu\text{g oil/gm wet weight mussel tissue}$ respectively.

Uptake constants for the three exposure concentrations were determined graphically by plotting $\mu\text{g oil/gm wet weight}$ against the relation $1 - e^{-k_2 t}$ (plot 1) or by plotting $\mu\text{g oil/gm wet weight}$ against $c_w t$ (plot 2). In plot 1, e is the base of natural logarithms; k_2 is the depuration rate constant in day^{-1} ; and t is time in days. In plot 2, c_w is the concentration of oil in the WSF in $\mu\text{g/ml}$ and t is time in days. Both types of uptake plots have unique and valuable features. In plot 1, the slope of the least squares linear regression line is $(k_1/k_2)(c_w)$ which can be

used to calculate k_1 , the uptake rate constant knowing k_2 and c_w . In addition, the slope in plot 1 is equal to the maximum concentration of oil that can be accumulated by the mussels. In plot 2, the slope of the least squares regression line is k_1 , the uptake rate constant. Plot 2 differs from plot 1 as it does not depend on knowledge of k_2 , the depuration rate constant. In addition, the exposure concentration, c_w , can be estimated directly from this graph if the exposure time, t , is known.

The least squares linear regression lines were statistically significant in both plot 1 and plot 2 for each of the three exposure concentrations using the uptake values for the six time points for each plot. For plot 1, the literature value of $k_2 = 0.15$ was used (Mason, 1988,1). The uptake constants were 222, 249, and 219 day^{-1} for plot 1 and 212, 238, and 209 day^{-1} for plot 2 for 0.1%, 1%, and 10% WSF exposures respectively. The maximum uptake values derived from the slopes of the linear regression lines for uptake values in $\mu\text{g oil/gm wet weight}$ plotted against $1 - e^{-k_2 t}$ (plot 1) were 6.3, 39, and 622 $\mu\text{g oil/gm wet weight}$ for 0.1%, 1%, and 10% WSF exposures respectively. These data confirm that tissue concentrations in mussels are directly related to the product of exposure concentration and time; the uptake constants are independent of exposure concentrations; and the two methods of plotting the uptake data are mathematically equivalent for determining the uptake constants.

The concentration of aromatic hydrocarbons (AHC) in the water associated with mussels taken from sites in Prince William Sound were estimated using our laboratory uptake curves. Tissue concentrations of AHC were determined in

triplicate by the Auke Bay Laboratory using GC and reported in Fish/Shellfish Study #11 (Injury to Herring). The mussels were retrieved between 4/28/89 and 5/4/89 from Fairmont Bay (5 transects); Naked Island (14 transects); Storey Island (2 transects); and Rocky Bay (4 transects).

The mean $\pm$ SD values for the tissue concentrations for the indicated number of transects for the four sites were 0.1047 ± 0.0404 , 0.6955 ± 0.6227 , 0.8445 ± 0.5518 , and 1.256 ± 0.995 μg AHC/gm wet weight, respectively. The corresponding mean $\pm$ SD values for μg AHC/liter (PPB) using our laboratory uptake curves converted to GC values for the 0.1% WSF exposure were 0.493 ± 0.190 (FB); 3.273 ± 2.930 (NI); 3.974 ± 2.597 (SI); and 5.911 ± 4.493 (RB), respectively, for a one day exposure. The elevated values for AHC in the water or in the mussels at the oiled sites relative to Fairmont Bay values were significant by the t-test ($p < 0.05$). Similar values were obtained from the uptake curve conducted with 1% WSF or 10% WSF.

The predicted concentrations of AHC/ml indicate that significant but low levels of oil pollution were present at selected sites in Prince William Sound 4-5 weeks after the oil spill. The estimated aromatic hydrocarbon concentrations in the water at the sites may be high as the predicted values rest on the assumption of a 24-hour site exposure. Longer exposure periods would proportionately de-

crease the AHD values in the water. In addition, the high fluorescence to GC conversion factor we determined (8.9) could exaggerate the predicted values. Finally, oil droplets trapped in the prepared WSF or at the environmental site would also exaggerate the values for AHC in the water. Subject to the confirmation of the oil concentrations in the mussels used for the uptake curve, the rapid and convenient laboratory assay described in this paper may provide a simple method for predicting oil exposure concentrations in water samples from determined oil concentrations in mussel tissue.

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Impacts to Intertidal Invertebrates in Herring Bay, Prince William Sound, Following the Exxon Valdez Oil Spill

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Intertidal monitoring and experimental studies were carried out in Herring Bay, Knight Island, Prince William Sound (60°N:147°W) as part of the Coastal Habitat Injury Assessment program (CHIA). Population densities for several species of invertebrates were compared between matched oiled and control sites from 1990 to 1992. Limpets were included because they are important intertidal grazers. The periwinkle, *Littorina sitkana*, the dog whelk, *Nucella* spp., and the six-armed starfish, *Leptasterias hexactis*, were studied because they lack a free-swimming larval stage, and may recover slowly after a large reduction in population over a large area.

The initial study design in Herring Bay was to select a range of sites with different oiling histories, including a non-oiled control site, and oiled sites that were mechanically treated (washed) and bioremediated. Ideally, this combination of sites would have been replicated several times to achieve statistical rigor. However, after surveys in May, 1990, review of data from the Exxon/Federal/State spring shoreline assessments, and detailed discussion with Alaska Department of Environmental Conservation monitors working in Herring Bay, treatment history of specific sites could not be determined with certainty. Therefore, the matched pair design of oiled and non-oiled study sites was adopted and treatment was not included as a variable. Sites were matched for substrate compo-

sition, slope, direction and solar aspect, wave exposure, and common biological communities. Control sites were restricted to the southeast corner of Herring Bay, where ice had prevented oil from entering in the spring of 1989. Most matched oiled sites were located in the lower-mid and western portion of Herring Bay.

In 1990, permanent 50 X 20 cm quadrats were established at five pairs of sites: three sheltered rocky and two sheltered coarse grained beaches. At each site six quadrats were located within each of three tide levels, or meters of vertical drop (MVD), below mean high high water. Within each quadrat, all limpets, *Nucella* spp., *Littorina sitkana* and *Leptasterias hexactis* were counted. In 1991, one additional protected rocky site pair and one coarse-textured oiled beach site, for comparison to an existing control site, were added to the study. In 1992, examination of these quadrats continued and four additional protected rocky site pairs were added. There were only three quadrats per MVD for the pairs added in 1992, but they were randomly established as previously done.

Densities of the limpet, *Tectura persona*, have remained significantly higher at control sites during the 3-year study period, with this difference pronounced at the 1 and 2 MVD ($p < 0.05$, at 5 of the 7 pairs at 1 MVD; $p < 0.01$ at 3 of 7 site pairs at 2 MVD, repeated measures ANOVA). Also, the differences in density of *T. per-*

sona between control and oiled sites tended to increase substantially in 1991. This pattern may have continued into 1992, but is uncertain with only two sample periods for this season.

In sheltered rocky habitats, *T. persona* is common in the upper intertidal but is not abundant at the third MVD. In contrast, *T. persona* is reasonably common at the third MVD in coarse textured habitats, and the density remains significantly lower at the oiled site in only one of the three pairs at this tide level ($P < 0.04$, repeated measures ANOVA). For the four site pairs added in 1992, the data support the general trends seen in the original sites. On both sample dates at the 1 MVD, *T. persona* densities were significantly lower ($p < 0.04$, t-Test) at two of the oiled sites, with weak significance ($p < 0.17$, t-Test) at the remaining pairs. A similar pattern occurred in the second and third MVD, but differences were not consistently significant.

Another limpet, *Lottia pelta*, is distributed lower on the shore, being more abundant in the second and third MVD. Similar to *T. persona*, differences in *L. pelta* densities between control and oiled sites peaked in 1991 for the second MVD at most sites. However, *L. pelta* appears to be recovering more rapidly than *T. persona*. *L. pelta* densities at the four sites established in 1992 were generally greater at control than oiled sites, but were significant in only a few cases.

The periwinkle, *Littorina sitkana*, was significantly less dense, especially at MVD 2, at four of the seven oiled sites compared to controls over the course of the study ($0.000 \Rightarrow p \leq 0.03$, repeated measures ANOVA). Recovery has been minimal. For the four sites added in

1992, *L. sitkana* densities tended to be lower at oiled than control sites though differences were not always significant. For example, significantly fewer individuals were found at oiled sites in only 1 of the site pairs at the 2 MVD in the spring but by summer, differences were significant at three of the four pairs ($p < 0.05$ t-Test).

The dog whelk, *Nucella lamellosa*, had only sufficient densities for statistical comparison at one of the seven site pairs and there were no differences found over the course of the study. However, density at the oiled site dropped by one-half toward the latter part of the 1990 season, and a similar drop was observed in 1991. However, *N. lamellosa* densities remained higher at the oiled site compared the control on both sample dates in 1992.

The other species with direct embryological development, *Nucella lima* and *Leptasterias hexactis*, were either not present or found in very low densities, and no differences were observed between site pairs.

Reductions in densities of invertebrates, particularly intertidal grazers such as limpets and periwinkles have been reported for previous oil spills (Nelson-Smith, 1977; Mann and Clark, 1978; Southward and Southward, 1978), and our findings are consistent with these earlier results. Effects from the Exxon Valdez oil spill on the invertebrates in Herring Bay have been variable, but damage has been documented, and recovery remains incomplete in some cases, especially the upper intertidal zone. The loss of *Fucus* from the 1 MVD is believed to be largely responsible for the inability of limpets to survive there. However, with the exception of *T. persona*, the other af-

affected invertebrates show populations increases since the spill.

A main hypothesis of the population studies was that brooding invertebrates, such as *Littorina sitkana*, *Nucella* spp. and *Leptasterias*, would suffer greater long term consequences because of limited dispersal, and the potential effect of oil on fecundity and development. Based on the data collected to date, only *L. sitkana* appears to have been slightly affected by oil. In 1990 *L. sitkana* showed significant differences only at the coarse-textured sites, but in subsequent seasons abundances have been reduced at the 1 and 2 MVD of oiled sites, similarly to the changes seen in limpets.

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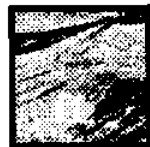
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Exxon Valdez Oil Spill: Recruitment on Oiled and Non-Oiled Substrates

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As part of the Coastal Habitat Injury Assessment program, intertidal experiments were established in Herring Bay, Knight Island, Prince William Sound (60° N: 147° W) in 1990. Two separate studies, continued through 1992, examined the effect of north slope crude oil on recruitment of barnacles, *Fucus* germlings and filamentous algae. The effects of oil on algal recruitment are reported in a separate abstract in this symposium.

In 1990 two oiled sites and two control sites of similar character were selected to study recruitment on tarred and clean vertical rock faces. At each site, paired 10 X 10 cm plots were randomly established. One member of each pair was scraped and brushed to remove all visible tar and/or barnacles. The sites were periodically visited and numbers of barnacles and *Fucus* germlings were noted. In 1991 the study was expanded to include a total of five site pairs and grazing-exclusion cages were added to half of the study plots.

The 1990 data show that barnacle recruitment was initially retarded on the oiled plots at oiled sites, compared to the scraped ones ($p < 0.05$, paired t-Test), but these differences began to fade at several sites over time ($p = > 0.2$, paired t-Test). This trend is most evident with the caged plots. The control sites have had consistently greater densities on the control plots compared to the scraped ones. *Fucus* germlings began to emerge at control sites in 1990, and only in low density at

the oiled sites in 1991. In 1992 *Fucus* was in greater abundance at oiled sites, and were greater on the unscraped plots (both control and oiled sites) compared to the scraped ones ($0.2 > p < 0.8$, paired t-Test).

Densities of grazers (limpets, *Littorina sitkana*, and *L. scutulata*), were significantly greater at control sites compared to oiled sites in 1991 ($p < 0.05$, ANOVA for four of five site pairs).

In 1992 data analysis was further subdivided to include measurement of surviving adults of *Semibalanus*, *Balanus*, and *Chthamalus dalli* from the previous seasons. Fewer differences were observed in 1992 compared to the previous years for all species and ages. Adult barnacles represent those individuals which have successfully recruited onto scraped plots. Adults are found in greater density at control sites because these were nonoiled individuals alive before the spill and left as controls. However, at the oiled sites, adults have successfully recruited on both plots. Within cages, the density of surviving adults are similar on both oiled and scraped plots. However, on uncaged plots, this pattern is reversed. *Chthamalus dalli* were generally greater on the scraped plots compared to the oiled/control plots.

In a second recruitment study, initiated in 1990, three pairs of oiled and control sites were selected for transplanting oiled substrates. The substrates used were rocks retrieved from an oiled shoreline in Herring Bay, as well as rocks treated with fresh North Slope crude oil,

taken from the *T/V Exxon Valdez* in 1989. Tarred rocks of similar size were collected from an oiled beach in Herring Bay, and represent a substrate coated with 1-year-old *Exxon Valdez* Prudhoe Bay crude oil (EV). One-half of each rock was cleaned with the solvent methylene chloride (MeCl_2) to remove the oil (Duncan, et al. 1992a). In addition, rocks approximating the sizes of the EV rocks were collected from a similar, but unoiled, beach. Half of each rock was dipped in fresh Prudhoe Bay Crude (PB) until a "tarred" coating was achieved. These rocks were allowed to dry for several weeks and were handled in a manner identical to the EV rocks.

As a control for possible effects of MeCl_2 on recruitment, half of six unoiled rocks were "cleaned" with MeCl_2 (one rock placed per site). As a control on surface heterogeneity, white clay tiles were included in the experiment as oiled and clean pairs.

At each of the six experimental sites, 12 EV, 12 PB rocks and 6 tile pairs were placed randomly at the elevation contour 2 m below MHHW (Mean Higher High Water, included in the upper intertidal zone). Control rocks for MeCl_2 were also placed at each site. Periodically, settlement by barnacles and macro algae on each surface type was recorded.

In 1991, the study was modified to include nine pairs of red clay tiles at each study site. Six of the pairs consisted of a tarred and a clean tile and half of these pairs had cages constructed around them to exclude grazers. The remaining three pairs consisted of a clean tile and a tile painted black (rather than oiled) as a control for dark coloration and possible temperature differences (Straughan, 1976). The tiles were also randomly placed at the 2-m contour.

Only 2 oiled sites were consistently colonized over the three year period (Sites 1322X & 1723X). Control sites received fewer recruits of all species (except for *Fucus*) compared to the oiled sites. The rocks deployed in 1990 were weathered and dislodged from the substrate over the course of the 1990-91 winter at most sites. Many of the rocks had the oil weathered completely from the sampling surface and, consequently, could not be sampled in 1991.

During 1990, the oiled halves of EV and PB rocks had lower densities of barnacle recruits in most cases. Initial settlement (early July) tended to be greater on unoiled halves but differences were only weakly significant ($p \leq 0.18$, paired t-Test). There was little recruitment in general later in the season. The freshly coated PB rocks showed greater differences, which were significant over several sample dates at both sites ($p \geq 0.001 < 0.1$, paired t-Test). However, the oil on the PB rocks was washed away in many cases and, by the end of the season with little recruitment occurring, no differences remained. The control rocks cleaned with methylene chloride had no differences in barnacle recruits between treated and untreated sides ($0.35 \Rightarrow p < 0.9$, paired t-Test).

The six tile pairs placed in the field in 1990 had fewer barnacle recruits. Densities were significantly lower ($p = 0.01$, paired t-Test) in early July, and weak significance ($p \geq 0.07 < 0.2$, paired t-Test) was found at 6 of 10 sample dates.

There was sparse settlement on the tile pairs placed at control sites in 1991. There were few differences in barnacle recruits, *Fucus* germlings and percent algal cover and no trends were apparent. However, the two sites (1322X and 1723X) which had good recruitment in 1990 also

had substantial recruitment activity in 1991. The caged tiles of both sites had somewhat greater numbers of barnacles on the unoiled tile compared to the oiled tile except toward the end of the recruitment season, but differences were only weakly significant ($p > 0.18$, paired t-Test). The painted tile pairs had similar levels of barnacle and *Fucus* recruitment on both tiles.

In 1992, barnacle recruitment was substantially lower than in 1991 and patterns were not evident. Data on adult barnacles present on the tiles indicate very few recruits reach adult sizes. *Chthamalus dalli* tended to recruit better on oiled tiles in caged treatments and on unoiled tiles in uncaged treatments, and were not significantly different on painted and unpainted tiles.

In 1991, *Fucus* did not recruit on uncaged or painted tile pairs and only began to recruit on caged, clean tiles at the end of the season. Again in 1992, there was almost no *Fucus* recruitment on uncaged or painted tile pairs. For caged treatments, recruitment tended to be higher on clean tiles with a slight tendency for higher recruitment at control sites.

Recruitment, including that of algal cover, appears to play the major role in structuring invertebrate communities at the Herring Bay study sites. These studies show that oil had an initial effect on barnacle recruitment, and depending upon substrate character, may have a moderate to long-term effect on algal recruitment (Duncan et al, 1992b). Within Herring Bay rock substrate differs from site to site, and several of the study sites have a more porous substrate than others.

For barnacles it is likely that residual

tar is an unstable settlement substrate and the reduced densities are a consequence of tar sloughing rather than toxicity. The sites showing the most rapid increases in invertebrate abundance are those most exposed to open water or tidal currents, which probably increases larval availability at those sites. Not surprisingly, these were also the sites hit by the floating oil. Although care was taken in matching rock sizes, the parent material (including the degree of porosity for each rock) varied greatly. This was especially true with the PB rocks selected. Many rocks were dense and non-porous and the oil quickly dissipated to the extent that comparisons between oiled and non-oiled halves could no longer be made.

The tile pairs were much less variable. Tiles placed in the field in 1990 were of a silicious clay and may have retained north slope crude more effectively than the red clay tiles used in 1991. Nonetheless, all tiles have retained a degree of oil staining not found on many of the rocks.

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The Response Process: The Goals of the Oil Spill Health Task Force

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The day of the *Exxon Valdez* oil spill had special meaning for the 4,500 Alaska Natives who live on the shores of Prince William Sound and depend upon the sea and shoreline to maintain a subsistence lifestyle. Walter Meganack, the tribal chief of the native village of Port Graham, described the day as "the time when the water died." He said that the native story was different from the white man's story.

"Our values are different, how we see the water and the land, the plants and the animals are different. What the white men do for sport and recreation and money, we do for life: for the life of our bodies, for the life of our spirits, and for the life of our ancient culture. The water is sacred." (Meganack, 1989).

The sea, shoreline and inland areas also provided the majority of protein intake in the diet of Alaskan Natives in the area.

Household surveys conducted by the Subsistence Division of the Department of Fish and Game prior to the oil spill from representative Prince William Sound villages documented up to 300 pounds of protein per person per year came from wild meat, fish and fowl (ADF&G, 1984). By comparison the comparable amount of the above food purchased in the Western U. S. is 220 lb.

The Oil Spill Health Task force was formed initially in April 1989 to respond in as timely and informative way as possible to the public's inquiry about the short and long term safety of subsistence food exposed to *Exxon Valdez* oil. The collective knowledge of the initial group

of members (Indian Health Service, State Division of Epidemiology, Subsistence Division of the Alaska Department of Fish and Game, The North Pacific Rim Native Corporation and the Kodiak Area Native Association) on the toxicology of oil and subsistence food was nonexistent.

Calls for federal assistance were not answered. The reason for the lack of response soon became clear. In a review of the world literature on behalf of the Oil Spill Task Force, toxicologist Dr. Gary Winston of Louisiana State University found "a virtual absence of literature which investigated any human health effects, concerns or implications as a function of oil spills other than those which were related directly to primary exposure to clean up crews." (Winston, 1990). Further inquiry to the Food and Drug Administration revealed no established safety levels for hydrocarbons for food.

The Task Force (joined in the late summer of 1989 by representatives from NOAA, the Toxicology Division of the Exxon Corporation and the Alaska Department of Environmental Conservation) developed the following mission: obtain as much historical information as possible about health effects of crude oil by both direct exposure and exposure through the food chain, conduct studies of subsistence foods for contamination with oil, interpret the results of these studies, and disseminate this information to the public.

As sample data on hydrocarbon concentrations in subsistence foods became available, expert advice was needed to interpret the results. Through NOAA's

help the task force was able to gather a group of nationally known senior toxicologists and scientists in what has become known as the Oil Spill Expert Toxicology Committee (Shank, 1990). Their recommendations formed the basis of our communication to the public.

At our request the FDA did a quantitative health risk assessment of subsistence food exposed to *Exxon Valdez* oil, and in August 1990 we received an advisory opinion of the safety of aromatic hydrocarbon residues found in subsistence foods most important to Alaskan Natives.

This then is the background to the monumental effort that ensued to insure food safety and the efforts of the Task Force to communicate this information to subsistence users in the numerous villages in the oil spill area.

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Subsistence Uses of Fish and Wildlife Resources in Areas Affected by the Exxon Valdez Oil Spill

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This paper discusses changes in subsistence uses of fish and wildlife resources in 15 predominantly Alaska Native communities whose hunting, fishing, and gathering areas were affected by the Exxon Valdez Oil Spill. It is based upon research conducted by the Division of Subsistence of the Alaska Department of Fish and Game. Study communities include Tatitlek and Chenega Bay in Prince William Sound; English Bay (Nanwalek) and Port Graham in lower Cook Inlet; Akhiok, Karluk, Larsen Bay, Old Harbor, Ouzinkie, and Port Lions in the Kodiak Island Borough; and Chignik, Chignik Lagoon, Chignik Lake, Perryville, and Ivanof Bay on the Alaska Peninsula. In 1990, the population of these 15 communities was 2,036, 82.3 percent of which was Alaska Native.

Prior to the spill, the division had conducted baseline research in all 15 villages. These studies found that subsistence harvests in these communities in the 1980s were large and diverse, ranging from about 200 pounds per person to over 600 pounds per person usable weight per year. These are substantial harvests, given that the average American family purchases about 222 pounds per person of meat, fish, and poultry annually. These subsistence harvests contained a wide variety of resources, including salmon and other fish, marine invertebrates, land mammals, marine mammals, birds and eggs, and wild plants. Virtually every household in all 15 villages used and harvested wild foods, which were widely shared within

and between communities. A patterned seasonal round of subsistence harvesting structured much of economic, social, and cultural activities in each community.

In early 1990, division researchers interviewed representatives of 403 households in these 15 communities. Study findings revealed that after the spill, subsistence harvests declined markedly in the 10 communities of Prince William Sound, lower Cook Inlet, and the Kodiak Island Borough compared to pre-spill averages. Annual per capita harvests in Chenega Bay and Tatitlek were down 57 percent. In these villages, the range of resources used also dropped in the 12 months after the spill. While the average household in Tatitlek used 22.6 kinds of wild foods from April 1988 through March 1989, in the next year, this average was only 11.6 types. The change at Chenega Bay was much like that of Tatitlek. In a 12 month study year in 1985-86, the average household at Chenega Bay used 19 kinds of wild foods, compared to just 8.2 kinds in the year after the spill.

Very similar changes were documented for English Bay and Port Graham. Compared to 1987, harvest quantities were down 51 percent at English Bay and 47 percent at Port Graham. The range of resources used per household dropped at English Bay from an average of 25.0 kinds in 1987 to 13.7 kinds in 1989. At Port Graham, the household average was 21.5 in 1987 and 11.2 in 1989.

Subsistence harvests in all six Kodiak

area villages also declined in 1989 compared to pre-spill averages, although a wider range of changes was documented. Harvests as measured in pounds per person per year were down 77 percent in Ouzinkie, 60 percent in Karluk, 52 percent in Port Lions, 40 percent in Old Harbor, 31 percent in Larsen Bay, and 12 percent in Akhiok.

In contrast, subsistence harvests in the five Alaska Peninsula communities showed little change, or increased, in 1989 compared to 1984, the only pre-spill year for which comprehensive data are available. Household interviews revealed that the presence of sheen, mousse, and tar balls temporarily disrupted subsistence harvests near these communities. However, most families resumed subsistence activities within several months. Resource use diversity also remained very high in the Alaska Peninsula villages, ranging from 15.3 resources per household in Chignik Lagoon to 29.7 kinds per household in Ivanof Bay.

When asked to provide reasons for declines in subsistence harvests in the year following the spill, 33.2 percent of the sampled households attributed reductions in overall subsistence harvests to concerns about resource contamination, and 44 percent said such a concern had caused a reduction in their harvest of at least one kind of subsistence food. Levels of concern about contamination were notably higher among Prince William Sound (92.1 percent) and Lower Cook Inlet (77.8 percent) households than in the Kodiak area (29.5 percent) or Alaska Peninsula (22.8 percent) communities. Other reasons cited for lowered levels of subsistence uses included the time harvesters spent on the oil spill cleanup and the perception that less resources were available because of spill-induced mor-

talities.

Regarding contamination concerns, it should be noted that little specific information was available to subsistence harvesters concerning the safety of using subsistence resources from the spill area until September 1989, by which time spring, summer, and most fall harvest opportunities had passed. Complete information from tests of fish and shellfish from subsistence foods testing programs were not available until February 1990, and information about marine mammals, birds, and deer was not available until June 1990. Interviews conducted in 1990 found that many households continued to express doubts about subsistence food safety. Respondents cited the relatively small number of samples tested in 1989, the limited sites examined, and the limited range of species tested as some reasons for their continuing questions. Also, hunters and fishermen continued to observe the presence of oil in harvest areas as well as dead and damaged wildlife which they attributed to the spill. Such signs were understood as evidence of continuing danger and many households thus acted cautiously with respect to using subsistence foods. Such behavior is culturally consistent for people whose survival has long relied upon their observations of the natural environment.

In 1991, the division conducted follow-up interviews with 221 households in seven spill-area villages pertaining to subsistence harvests during the second post-spill year. Harvest levels increased at Port Graham, Larsen Bay, and Karluk, and matched at least one pre-spill measurement. The range of resources used was also up substantially in all three communities. In two other communities, Ouzinkie and English Bay, harvest levels also increased, but

remained below pre-spill averages. This general increase in harvest levels and range of wild foods used suggests some renewed confidence in using subsistence foods during 1990.

On the other hand, lingering concerns about food safety were expressed in all five villages. Some families reported that they resumed their subsistence harvests despite misgivings because they could not afford to purchase substitutes and could no longer do without culturally important foods.

In contrast, no evidence of a recovery in subsistence uses in the second post-spill year was found for Chenega Bay and Tatitlek. At Chenega Bay, subsistence harvests from April 1990 through March 1991 were 139.2 pounds per person, virtually the same as the previous

year (148.1 pounds per person) and still well below the pre-spill average of 340 pounds per person. At Tatitlek, the 1990-91 per capita harvest was 152.0 pounds, compared to 214.8 pounds per person in the first post-spill year and a pre-spill average of 497.6 pounds per person.

In these Prince William Sound communities, deep concerns about the safety of using subsistence foods from their traditional harvest areas continued. In addition, respondents from Chenega Bay and Tatitlek reported perceived declines in the numbers of some important subsistence resources, such as certain species of waterfowl, marine invertebrates, and marine mammals, which led to well below normal subsistence harvests during 1990-1991.



Overview of Subsistence Food Safety Testing Program

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The Oil Spill Health Task Force, a group chaired by a Public Health Service physician and composed of representatives from state and Federal agencies, native organizations, and Exxon, was formed soon after the *Exxon Valdez* oil spill to address public health concerns resulting from the spill. The Oil Spill Health Task Force served as a forum for design of the subsistence food safety testing study and the communication of the results and conclusions derived from the study to the subsistence communities.

The main objectives of the study were to determine if subsistence foods were contaminated as a result of the spill and to assess the implications for subsistence food safety. In order to put together a comprehensive study, NOAA and Exxon signed a Memorandum of Understanding in the summer of 1989 which had three principal features: (1) NOAA biologists would accompany Exxon-sponsored collection teams to the villages and participate in the sample collections; (2) tissue samples from fish and shellfish subsistence resources would be analyzed for aromatic contaminants by the NOAA National Marine Fisheries Service Environmental Conservation Division laboratory in Seattle; and, (3) all data from the study would be made public.

These steps were taken to ensure that there were no questions about the validity of the data produced by the study and that the results and the implications for human health were made available to public health officials and the communities in a timely manner.

This paper addresses the general sampling strategy, the numbers and types of samples collected, the identification of the most contaminated sites, and issues related to the interpretation of the shellfish results. Subsequent papers in this session will address in more detail the results from chemical analysis of shellfish, fish, and marine mammal tissues, the implications of those results for human health, and an evaluation of the methods used in risk communication.

The study area included subsistence seafood collection areas in Prince William Sound (Tatitlek and Chenega Bay); Lower Cook Inlet (including the Outer Kenai Peninsula villages of Port Graham and English Bay, and Windy Bay); and Kodiak Island (Kodiak City, Chiniak, Larsen Bay, Karluk, Akhiok, Old Harbor, Ouzinkie, and Port Lions). Four subsistence areas on the Alaska Peninsula (Chignik, Perryville, Ivanof Bay, and Kashvik Bay) were sampled by ADF&G in 1990. Reference samples were collected from near the village of Angoon in 1989 and Yakutat in 1990. The initial sampling plan called for approximately equal numbers of samples from each village area, although there were considerable differences in the potential degree of impact from the *Exxon Valdez* oil. Sampling sites that represented important subsistence use areas were selected in consultation with village representatives. The degree of oiling was not a major factor in site selection. Two important subsistence beaches for the collection of intertidal shellfish were identified in each

village area. Target species included several species of intertidal shellfish (mussels, clams, chitons), bottomfish (primarily halibut), and salmon. Other species were also collected but in much smaller numbers. Intertidal stations for the collection of shellfish did not represent exact locations on a given beach. Thus, several species (e.g., mussels, clams, and chitons) were often collected from the same station, although each species occupied a different habitat (tidal elevation, substrate) in the beach.

In 1990, the number of sample stations and the number of samples per station were increased. In 1991, only shellfish were collected. The focus in 1991 was in southwestern Prince William Sound near Chenega Bay, because that part of the Sound had potentially the greatest amount of oiled shoreline, and Windy Bay, because that was the most heavily-oiled site sampled and was a good site to examine changes in concentration over time. Both Exxon and the Alaska Department of Fish and Game sponsored sample collections and analy-

ses in 1990 and 1991.

A total of over 1,000 shellfish samples collected from more than 65 stations were analyzed for aromatic contaminants in three years of sample collections. Five stations had one or more shellfish samples with total aromatic contaminant concentrations exceeding 1 ppm (part per million): two stations in Prince William Sound (one in Port Ashton in Sawmill Bay near the village of Chenega Bay and the other on southwestern Elrington Island); two islands in Windy Bay on the Outer Kenai Peninsula; and a station on Near Island, near the Kodiak boat harbor. The Windy Bay stations and the Elrington Island station both had obvious visual evidence of oiling.

At the most contaminated station, an island in Windy Bay, total aromatic contaminant concentrations in mussels varied by as much as three orders of magnitude from the same collection period over a relatively small beach area. The highest concentrations were found in mussels collected at a location higher in the intertidal area.



Assessment of Exposure of Subsistence Fish Species to Aromatic Compounds Following the Exxon Valdez Oil Spill

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On March 24, 1989, the *Exxon Valdez* ran aground on Bligh Reef spilling Prudhoe Bay crude oil (PBCO) into Prince William Sound, a relatively pristine area in Alaska. The oil spread through the Sound and into coastal areas along the Gulf of Alaska, affecting fishing grounds used by Alaska Native villages. The Alaska Natives were concerned about possible contamination by the spilled petroleum of fish and shellfish used for subsistence.

In response to this concern the National Oceanic and Atmospheric Administration (NOAA) entered into an agreement with the Exxon Corporation and the Alaska Department of Fish and Game (ADF&G) to survey native subsistence fisheries for exposure of fish and shellfish to oil. Here, we present the results from chemical analyses of several species of salmon and bottomfish collected from a number of fishing grounds used by native villagers to assess the extent of exposure of fish to spilled oil and the accumulation of petroleum-related compounds in edible flesh.

Of particular concern was the accumulation of aromatic compounds (ACs) present in oil, because some of these ACs are suspected of being toxic to humans (Dipple et al., 1984). The ACs measured included polycyclic aromatic hydrocarbons and sulfur-containing ACs. Additionally, several of the ACs (e.g., alkylated naphthalenes, phenanthrenes and dibenzothiophenes) measured were char-

acteristic of oil and particularly PBCO (Krahn et al., 1992). The results from the analyses of edible flesh of fish, in addition to results from the analysis of shellfish (Brown et al., this volume), were then evaluated by the Alaska Oil Spill Health Task Force and the U.S. Food and Drug Administration (FDA) to assess the potential risk to Alaska Natives of eating seafood from their traditional fishing sites following the spill.

Previous studies (Varanasi et al., 1989) have shown that fish have the capacity to biotransform many ACs to polar metabolites that are readily excreted into bile, which is retained in the gall bladder. Moreover, these studies showed that the extensive biotransformation of ACs by fish greatly limits the accumulation of these compounds or their metabolites in edible tissues. Accordingly, a rapid and sensitive semiquantitative method (Krahn et al., 1986) for measuring concentrations of fluorescent aromatic compounds (FACs) in bile was used to estimate exposure of fish to ACs that are present in oil. These results were then used for prioritization of edible flesh samples for a more extensive analysis of the presence of parent ACs by gas chromatography/mass spectrometry (GC/MS).

Concentrations of FACs in the bile of five species of bottomfish and five species of salmon collected from oil-impacted sites located in Prince William Sound and along the Gulf of Alaska were

determined. Elevated concentrations of FACs in bile of fish from several sites indicated exposure to ACs. For example, in 1989, mean concentrations of FACs were highest in bile of both salmon and bottomfish from sites near Chenega Bay (6000 ± 3400 and 5100 ± 4000 ng phenanthrene equivalents per mg bile protein, respectively) and Kodiak (2800 ± 1900 and 7550 ng phenanthrene equivalents per mg bile protein, respectively), which were heavily impacted by spilled oil. Moreover, these concentrations were substantially higher than the concentrations in salmon (1100 ± 2200 ng phenanthrene equivalents per mg bile protein) and bottomfish (310 ± 120 ng phenanthrene equivalents per mg bile protein) from Angoon, a reference site.

In the following year, salmon and bottomfish from Chenega Bay showed marked decreases in biliary FAC concentrations (550 ± 250 and 1300 ± 850 ng phenanthrene equivalents per mg bile protein, respectively) suggesting that the level of exposure to ACs in salmon from the Chenega Bay site had declined. Interestingly, no temporal changes were evident in salmon sampled from sites near Kodiak in 1990 (no bottomfish were sampled from this site). The concentration (2400 ± 2000 ng phenanthrene equivalents per mg bile protein) of FACs in salmon from Kodiak was comparable to the concentration in 1989 indicating that exposure to ACs was unchanged for salmon captured at this site. These results and data from analyses of shellfish from Kodiak (Brown et al., this volume) suggest exposure to a source of ACs other than spilled oil.

The HPLC analysis of bile for FACs provides a rapid, semiquantitative assessment of exposure to ACs that are

present in oil but does not allow identification of individual compounds found in bile. In a related study (Krahn et al., 1992) conducted as part of the Natural Resources Damage Assessment effort, it was shown that the bile method is a sensitive indicator of exposure of fish to specific ACs present in oil. In the study of Krahn et al. (1992), metabolites of ACs characteristic of PBCO were identified, by GC/MS, in high proportions in bile of fish injected with PBCO and in fish sampled from sites in Prince William Sound shortly after the spill. Metabolites of these ACs were not present in bile of fish sampled from a reference site distant from the spill. These findings substantiated the use of the bile method to rapidly assess exposure of fish to ACs in the present study.

As mentioned, the major use of the results on concentrations of biliary FACs was to prioritize corresponding edible flesh samples for more extensive analysis of the presence of parent ACs by GC/MS. Samples showing a range in exposure to oil were chosen for comprehensive analysis to determine if fish were accumulating significant levels of parent ACs in their edible tissue and to substantiate that low concentrations of FACs in bile accurately reflect minimal exposure to ACs and, hence, little potential for accumulation of ACs in muscle tissue.

The concentrations of biliary FACs in fish from which samples of muscle tissue were analyzed for parent ACs ranged from 10 to 18,000 ng phenanthrene equivalents per mg bile protein. In contrast, concentrations of total ACs (sum of individual selected ACs) analyzed by GC/MS in corresponding muscle tissue were low. No appreciable concentrations [<1 ng/g wet weight (ppb)] of parent ACs were detected in muscle of

bottomfish, and although concentrations of ACs in muscle of salmon were somewhat higher they very rarely exceeded 100 ppb, and in the majority of samples the concentrations of ACs were less than 30 ppb.

These findings showed that exposure of salmon and bottomfish to ACs does not lead to any significant accumulation of parent ACs in edible muscle tissue. These results also corroborate findings from laboratory studies on the pathways of metabolism and disposition of ACs in fish, showing that polar metabolites of ACs accumulate in bile and that parent ACs do not accumulate to significant concentrations in muscle tissue (Stein et al., 1987; Varanasi et al., 1989). The consistency of the results from this field study and those of previous laboratory studies provided evidence of minimal risk of exposure of humans to parent ACs from consumption of edible flesh of fish captured at sites impacted by the spilled oil.

The extensive biotransformation by fish of ACs to polar metabolites raised the issue of whether metabolites were present at elevated concentrations in muscle tissue of fish exposed to ACs. Accordingly, a method similar to the bile screening method is being developed (Krone et al. 1992) to estimate concentrations of AC metabolites in fish tissues. This new method has not been validated as extensively as the bile method, thus the results must be considered preliminary. However, analysis of a few selected samples of muscle from salmon indicated that concentrations of AC metabolites were also quite low. This finding is consistent with previous laboratory studies (Stein et al. 1984) with ACs that showed metabolites of ACs as well as parent

compounds are low in muscle of fish, and demonstrates the importance of laboratory studies, which lead to the development of techniques that provide the necessary information to respond effectively to environmental emergencies.

In summary, the results of this survey of Alaska Native subsistence seafood provided substantial evidence that fish were exposed to ACs that appeared to originate, in most cases, from the spilled oil. However, edible muscle tissue in salmon and bottomfish were found to have concentrations of ACs that rarely exceeded the relatively low concentration of 30 ppb. These data showing low concentrations of parent ACs in muscle of fish exhibiting a substantial range in exposure to ACs were used by the Alaska Oil Spill Health Task Force and the FDA in arriving at an advisory position that consumption of the flesh of fish posed minimal risk to native Alaskans.

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Petroleum Hydrocarbons in Alaskan Invertebrate Subsistence Foods Following the 1989 Exxon Valdez Oil Spill

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The *Exxon Valdez* ran aground on Bligh Reef, Prince William Sound, Alaska on 24 March, 1989, spilling millions of gallons of Prudhoe Bay crude oil (PBCO). During the weeks following the spill, large amounts of oil flowed toward southwestern Prince William Sound, and as a result, many shorelines were oiled. The spreading of spilled oil raised many concerns, and among them was the concern that subsistence seafoods (e.g., fish and invertebrates) were contaminated (see abstracts by J. Field and J. Stein et al.) by the spilled petroleum.

The objectives of this paper are to describe the extent and duration of contamination of invertebrates by the spilled oil. The data from this study together with those described by Stein et al. (in this session) were used by the Alaskan Oil Spill Health Task Force and the U.S. Food and Drug Administration to address the issue of potential risk to native Alaskans of eating seafood from their traditional collection sites.

Crude petroleum contains many hundreds of individual chemicals which are generally divided into classes such as aliphatic hydrocarbons, aromatic hydrocarbons and other constituents containing sulfur, oxygen and nitrogen (Clark and Brown, 1977). Among these compounds are dibenzothiophene and alkyl-substituted dibenzothiophenes, which are significant components of PBCO and can serve as useful markers in discerning

PBCO. Among the constituents of petroleum, the aromatic hydrocarbons are considered the most toxic. The lower molecular weight aromatic compounds (LACs), those that contain three or fewer benzene rings, have been found to be the more acutely toxic, whereas the higher molecular weight aromatic compounds (four to six rings, HACs) are generally considered the more chronically toxic components.

A majority (99%) of the ACs in fresh PBCO measured in this study consisted of LACs. However, as the oil weathered, the concentrations of many of the more volatile and water soluble LACs decreased rapidly. Sixteen non-substituted ACs and 17 groups of alkyl-substituted ACs comprising some 270 individual chemicals present in PBCO were determined in subsistence seafood samples to assess the extent of contamination, to evaluate temporal changes, and to compare to possible sources.

More than a thousand samples of fish, marine mammals, and invertebrates were collected between 1989 and 1991 and were analyzed for ACs by gas chromatography/mass spectrometry according to Krahn et al. (1988). These samples included a variety of invertebrates from 80 sampling stations used by the residents of 17 villages for collecting subsistence seafoods. The focus of this paper will be on the four target invertebrate taxa; mussels (*Mytilus edulis*), butter clams

(*Saxidomus giganteus*), littleneck clams (*Protothaca staminea*), and chitons (order Neoloricata).

For discussion purposes, invertebrate samples (wet weight basis) are divided into four categories on the basis of the sum of concentrations of the ACs:

1. Not contaminated- <10 ng/g;
2. Minimally contaminated- from 10 ng/g to 100 ng/g;
3. Moderately contaminated—from 100 ng/g to 1000 ng/g;
4. Heavily contaminated- >1000 ng/g.

Invertebrates from most of the 17 villages and the two reference areas, Angoon and Yakutat, fell into the first two categories: not contaminated or minimally contaminated. Invertebrates from some stations at two sampling areas, Chenega Bay and Windy Bay, and from two stations near Kodiak Island were moderately or heavily contaminated with ACs.

Following the spill, oil was observed in Chenega Bay in the vicinity of station CHE1 (the beach below the village of Chenga Bay—used by villagers as a harvesting area). A tar mat about 1 m wide extended the length of the beach at the high tide line at CHE10 (the southern end of Elrington Island). Invertebrates from eight of eleven stations at Chenega Bay were not contaminated or were minimally contaminated. Invertebrates from three stations at Chenega Bay from one or more sampling events were moderately or heavily contaminated. Specifically, the mussels collected from CHE1 (1989) and CHE7 (at Port Ashton-1990) were moderately contaminated and those from CHE10 (1990) were heavily contaminated.

However, mussel samples collected from CHE1 in 1990 and 1991, and mussels collected from CHE7 and CHE10 in 1991 were minimally contaminated.

Butter clams collected from CHE7 (1990 and 1991) and littleneck clams collected in 1990 were moderately contaminated. Butter clams from CHE1 (1989, 1990, and 1991) and from CHE10 (1990) were minimally contaminated as were littleneck clams from CHE1 (1990 and 1991) and from CHE10 (1990, 1991). Butter clams from CHE10 (1991) and chitons from several stations were not contaminated.

Based on mean concentrations of ACs, invertebrate samples from two of the five stations sampled at Windy Bay (stations WNB4 and WNB5) were not contaminated or were minimally contaminated. Two stations, WNB1 and WNB3, located on two adjacent small islands in the mouth of Windy Bay, were both observed to be moderately to heavily oiled.

The degree of contamination of invertebrates from these two stations varied with sampling year and by species. Specifically, mussels from Windy Bay station WNB1 (1989) and from WNB3 (1990) were heavily contaminated, whereas the concentrations of ACs in mussels from these stations in 1991 were much lower (minimally to moderately contaminated).

Mussels were not collected at WNB1 in 1990 or at WNB3 in 1989. Only one sample was collected at WNB2, littleneck clams sampled in 1989, which were almost in the heavily contaminated category, with 960 ng/g ACs.

Invertebrate samples from two sampling stations on Kodiak Island (KOD3 and OHA4) also contained elevated concentrations of ACs. Station KOD3 was located near the city of Kodiak and adjacent to the boat harbor. An oily sheen was observed on one occasion at KOD3 by the field party while digging for clams. Mussels, butter clams, littleneck clams, and chitons from station KOD3 from both

1989 and 1990 were moderately or heavily contaminated.

Station OHA4 at the village of Old Harbor was also near a boat harbor. Oil was not observed to be present during sample collection at this site. Butter clams and littleneck clams collected at Old Harbor station OHA4 in 1989 and mussels collected in 1990 were just within the moderately contaminated category. Five of the six mussel samples and three of six butter clams (collected in 1989) were minimally contaminated.

The relative amounts of the ACs can be useful in evaluating temporal changes and to compare to potential sources. The relative amounts of the parent and alkyl-substituted ACs are useful for comparing patterns of these ACs among petroleum, petroleum products, related materials, and ACs in invertebrate samples. Dibenzothiophene and alkyl-substituted dibenzothiophenes (which comprise 27 discernible components in PBCO) were particularly useful in making comparisons.

For example, these chemicals were mostly absent in a sample of Cook Inlet crude oil but were much more prominent in a sample of Kuwaiti crude oil than PBCO. Graphs of these patterns from various oils and from certain moderately and highly contaminated invertebrate samples were compared and the same data were treated using principal components analysis and hierarchical cluster analysis. The relative amounts of these analytes in oil collected nine days after the *Exxon Valdez* spill were similar to those in the fresh oil except for a loss of the more volatile and water soluble components, particularly, naphthalene, C1- and C2-naphthalenes.

The pattern of ACs in a sample of oil collected from the beach at Snug Harbor

15 months after the spill was similar to that of the nine-day weathered oil. The patterns of ACs in mussels from Windy Bay station WNB3 collected in 1989 were similar to the oil sample from Snug Harbor except for additional losses of the more volatile and water soluble components of LACs. The temporal changes of the patterns of ACs in mussels collected from Windy Bay from 1989 to 1991 showed increasing losses of the more volatile and water soluble components of LACs. Evaluation of the patterns of ACs from mussel samples from CHE7 implies the presence of PBCO plus ACs indicative of combustion products.

ACs present in samples from Kodiak Island stations OHA4 and KOD3 were indicative of petroleum or petroleum products and the pattern of ACs was very similar to that of the nine day weathered PBCO. The pattern of the ACs in a mussel sample from KOD3 was essentially the same as in butter clams from the same site collected in March 1990, April 1990, and September 1990, and in a littleneck clam sample collected April 1990 at KOD3. There was little evidence of temporal change in the pattern of ACs in these clam samples from KOD3 which could imply a continual exposure to the same source of ACs. It is possible that there is a local and continual source of hydrocarbons in the area of these two stations that would have a pattern of ACs similar to the somewhat weathered PBCO.

In summary, invertebrate samples from most stations, including those from the two reference areas, were not contaminated or were only minimally contaminated with ACs typical of petroleum. Invertebrates from a few stations in the Chenega Bay sampling area and Windy Bay were moderately or heavily

contaminated with ACs. Invertebrates from one station (KOD3) on Kodiak Island were moderately to heavily contaminated with ACs and the concentrations did not appear to decrease significantly from 1989 to 1990 (samples were not collected in 1991) suggesting the presence of other possible sources of ACs not related to the *Exxon Valdez* spill. Mean concentrations of ACs in mussels from the more contaminated sites at Chenega Bay and Windy Bay generally decreased with time. However, because of sample

variability, it is difficult to draw conclusions about temporal trends.

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Investigations of Crude Oil Contamination in Intertidal Archaeological Sites Around the Gulf of Alaska

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The State of Alaska initiated a brief study during 1991 aimed at investigating the presence of crude oil in archaeological sites which occur in the intertidal zone. The concern was that cultural deposits subjected to oiling retained crude oil contaminants that would affect radiocarbon dates obtained from the sites. Designed to complement a larger, area-wide study funded through the U.S. Forest Service, the State study concentrated on 13 intertidal sites suspected to contain intact cultural deposits. Site selection was based on documented presence of cultural remains in the intertidal zone and evidence of beach oiling during the *Exxon Valdez* oil spill. Preliminary field examinations eliminated 10 sites from further examination because they lacked intact cultural deposits. One site above the high tide line was added to test presence of wind or storm wave-borne oil contaminants.

Laboratory studies about the effect of incorporation of crude oil on radiocarbon dating with datable samples suggested that significant skewing of dates occurred (Mifflin and Associates, 1991). The goals of the project therefore were to test the selected sites for presence of oil and to concurrently test the reliability of radiocarbon dates from the deposits. Comparison of time diagnostic artifacts from the tested sites with similar artifacts from well dated nearby sites was the method chosen to check validity of the oiled site dates. Sites were partially

excavated to obtain large enough collections and adequate radiocarbon samples to accomplish the comparison.

Sites were tested to reveal stratigraphy of deposits in the middle to upper intertidal zone and sediment samples collected to test for presence of subsurface oil. Selected sediment samples were submitted to the National Oceanic and Atmospheric Administration/National Marine Fisheries Service laboratory in Seattle for analysis by high performance liquid chromatography with ultraviolet detection (HPLC/UV). One sample from the AFG-098 Site on Shuyak Island and a sample from the SEL-215 Site on Nuka Island contained minute traces of petroleum hydrocarbons.

The AFG-098 Site on Shuyak Island contains cultural deposits which yielded artifact collections typical of Koniag Phase from the region and provides the best opportunity to test the hypothesis that contamination affected radiocarbon dating. Two cultural levels are clearly separate in the stratigraphic profile of the site. The Lower Component is an early Koniag Phase collection containing stemmed, ground slate, projectile points with barbs. Cross-section of the points is generally a flattened biconvex form although some points have a medial ridge on one face. Another form of point, triangular in outline with a flat basal facet, was found in the lower level of the site. Ground slate ulus from the component have straight, bifacially ground cut-

ting edges. The collection contains planing adzes of the variety usually inset into a bone socket and then hafted. Other diagnostic artifacts in the Lower Component include a slate fragment with an etched face, a stone saw, and chipped slate preforms for subsequent grinding into projectile points.

Similar, early Koniag Phase collections at well dated sites in the region have assigned ages of between A.D. 1350 and A.D. 1500. Age estimates for the Koniag Phase in the Kodiak Archipelago cite a beginning around A.D. 1100 to A.D. 1200. Age of the incised slate figurines or faces is particularly well defined during the A.D. 1350 to A.D. 1500 period. Triangular ground slate points with a flat ground basal facet typically date from A.D. 1300 to A.D. 1500. Four radiocarbon samples stratigraphically associated with the Lower Component range from A.D. 787 ± 110 to A.D. 1143 ± 65 . The four dates are acceptably within the earliest estimated range of the Koniag Phase and may indicate that the developmental stage of the phase occurred earliest in the northern Kodiak area.

The AFG-098 Upper Component collection provided an artifact collection which is clearly related to Koniag Phase materials elsewhere. Ground slate projectile points of several forms were recovered. Stemmed points, some with barbs and some with rounded shoulders, were the most common forms recovered. Medial ridges were present on some barbed forms but a flattened biconvex cross-section was most common among the stemmed points. A very distinctive ground slate projectile point with a triangular outline and a sharply defined flute or butt facet was found in the Upper Component. Several ground slate ulu forms occur in the component. Both

are bifacially ground and have straight to slightly convex cutting edges. One form has a distinctive notch or offset at the back of the blade which suggests hafting at the back edge near one end with the other end extending out of the handle. Other stone artifacts from the Upper Component include a stone saw, sawn slate fragments, whetstones, planing adzes, a quartz crystal, a hematite nodule for fire starting, and chipped slate preforms. Fragments of several bone or ivory dart heads with unilateral barbs, a barbed hook fragment, and bone awls are some of the organic artifacts recovered. A small jet labret was also found. Fragile artifacts carved from spruce wood, bark and grass matting, and fragments of a birch bark container were recovered from the saturated deposits. More than a dozen species of seeds were found in floor deposits of the Upper Component.

Comparison of the Upper Component collection with Koniag Phase or related site collections on Kodiak Island, both sides of the Alaska Peninsula, and the outer coast of the Kenai Peninsula demonstrate close similarities. The triangular point form with a basal flute consistently occurs in Koniag Phase or related collections dating from between A.D. 1500 and Historic times (A.D. 1750-1800). Other point and ulu forms are consistent with later Koniag Phase ages as well. Six radiocarbon dates obtained from the Upper Component levels of AFG-098 range from A.D. 1343 ± 60 to A.D. 1490 ± 125 .

The AFG-082 Site is an upland site eroding slowly into the intertidal zone which was tested for contamination from storm wave or wind deposited oil. The ground slate ulus, chipped bifaces, planing adze, and small single chamber

houses compare reasonably well with middle age range Kachemak Tradition remains, approximately 2,000 years old. The ratio of ground slate to chipped stone remains also supports a Kachemak Tradition comparison. Unfortunately, the AFG-082 collection is too small and typologically limited to provide a very accurate age estimate.

Two radiocarbon samples obtained from the site date to A.D. 203 $\pm$ 65 and A.D. 288 $\pm$ 65. The AFG-082 dates fall well within the expected age range determined from artifact typology. No evidence of oil was found in the site and no sediment samples were submitted for analysis.

The SEL-215 Site on Nuka Island contains intact cultural remains within a peat deposit in the intertidal zone. Time specific traits in the collection are meager but grooved splitting adzes suggest an age between A.D. 1000 and A.D. 1500. Inclusion of a glass trade bead in the collection is interpreted as an intrusive historic element.

Radiocarbon samples from the site yielded seven dates ranging from A.D. 1142 $\pm$ 60 to A.D. 1442 $\pm$ 105. A trace of oil was detected in one of the two sediment samples submitted for HPLV/UV screening. However, the radiocarbon determinations compare well with the expected age of the deposits.

The SEW-068 Site consists of a peaty intertidal deposit containing cultural remains which relate to Kachemak Tradition collections elsewhere. Grooved splitting adzes located nearby may not be associated however such tools have been dated to that early time from other sites in Prince William Sound. A general estimate of age for the peaty deposits, based on artifact typology, is 1000-2000 years ago. Geological age estimates based on

rates of isostatic rebound from the 1964 Earthquake and from long term regional subsidence indicate the deposits should be at least 1500 years old. Two wood samples from the saturated deposits provided radiocarbon dates of A.D. 10 $\pm$ 65 and A.D. 391 $\pm$ 65. The ages obtained from the culturally modified wood fragments agree roughly with the expected age of the cultural deposits.

Conclusions drawn from this study include:

- (1) Intertidal archaeological deposits at three of the sites investigated demonstrate that useful and important information is preserved in some intertidal sites even though there is sometimes no surface evidence of buried remains.
- (2) Traces of petroleum hydrocarbons in subsurface remains do occur although the origin of the contaminants in the sites tested is unknown.
- (3) Reliable radiocarbon dates can be obtained from oiled deposits. It is uncertain, however, whether that results in the tested sites from cleaning of the samples or simply lack of actual contamination. Examination of samples in the radiocarbon laboratory for oil contaminants should be routine and cleaning methods should be modified if necessary to remove identified contaminants.
- (4) Dating of intertidal or even exposed upland archaeological remains needs to involve every possible approach to dating, not just reliance on a single method. Archaeologists investigating chronological questions in the area of the *Exxon Valdez* oil spill need to be especially critical with their conclusions. No evidence was found in the sites examined that the *Exxon Valdez*

oil spill adversely affected the radiocarbon dating results. Damage to sites appears to be from erosion or vandalism rather than direct oiling.

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Generating Damage Restoration Costs for Archaeological Injuries of the Exxon Valdez Oil Spill

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This paper summarizes the results of a monetary assessment of damage for injuries to archaeological sites documented in the Exxon Valdez oil spill response records. Injuries attributable to the oil spill at 35 archaeological sites in Prince William Sound and the Gulf of Alaska were analyzed to estimate restoration costs for use by the Exxon Valdez Trustee Council in planning restoration of damages for archaeological resources. The damage assessment was accomplished in two steps.

First, a damage assessment panel, chaired by the author, met to consider restoration costs based on the archaeological injury data. Second, working from the findings of the panel, additional analyses were conducted by the author to calculate specific restoration costs for the archaeological injuries. Two levels of restoration costs were produced by the damage assessment process. First, site-specific restoration costs were developed for the archaeological sites identified as having substantive injuries. Second, gross restoration costs were estimated for projected numbers of sites injured by the oil spill.

The procedures employed by the damage assessment panel were carried out in nine steps:

1. The data on the 35 archaeological sites with injuries attributable to the oil spill and other relevant data sources were reviewed.
2. A conceptual framework on which restoration costs would be based was developed from the damage assessment model contained in the Archaeological Resources Protection Act of 1979, as amended (ARPA).
3. Documented oil spill injuries to archaeological sites were analyzed and grouped into two major categories, those resulting from oiling and those resulting from oil spill response activities.
4. Two restoration options were developed, one for ten years of oil effect monitoring and one for direct physical restoration. These were used to formulate specific restoration measures appropriate for the two categories of injuries.
5. Ten injured sites were eliminated from further consideration because appropriate site restoration work had already been accomplished or the site damage was not severe enough to require restoration. Damage at one site was determined to be unrestorable.
6. Three categories of site specific restoration proposals were developed for the remaining 24 sites: sites recommended for monitoring only, sites recommended for direct physical restoration only, and sites with both types of measures recommended.
7. Standard levels of effort were formulated for direct physical restoration in year one and in years two through ten for oil effect monitoring. Appropriate site-specific salary estimates

were generated using standard levels of effort as a guide.

8. Site-specific support costs necessary to carry out direct physical restoration measures and oil effect monitoring were estimated.
9. Restoration costs were estimated for four different injury scenarios, each with a different number of injured sites, using average per site costs for direct physical restoration measures and oil-effect monitoring.

Two principal sets of findings were produced. The first consists of the proposals for site restoration measures and costs for the 24 archaeological sites with substantive injuries. The second consists of restoration costs for projected numbers of sites injured by the oil spill.

Site-specific restoration proposals are based on the oil effect monitoring and direct physical restoration measures. Using the above criteria, there are five sites in the oil effect monitoring only category, 14 sites in the direct physical restoration only category, and five sites at which both types of measures are recommended.

The first element of the restoration cost proposals for the 24 sites are personnel salaries. The salary figures for 17 of the 24 sites are based on the three standard levels of effort defined above. Due to special circumstances, seven sites have salary figures based on variations of the standard computations.

Five sites have cost proposals involving only salaries for ten years of monitoring. The standard salary figure for year one oil effect monitoring is \$2,904.51, and \$2,202.17 per year for years two through ten or \$19,819.53 for 9 years. Therefore, the total salary figure for each of these sites is \$22,724.04.

Salary for direct physical restoration

only is proposed for 11 sites. The total salary figure for each of these sites is \$3,155.79.

Only one site is in the oil effect monitoring and direct physical restoration salary category. The total salary figure for this site consists of the standard salaries for ten years oil-effect monitoring, plus costs for direct physical restoration—\$25,879.83.

Four sites have salary figures for direct physical restoration measures above the standard level of effort, as well as oil effect monitoring. One has disinterred human remains which required the addition of eight days of project supervisor time for consultation with Native Corporations. The result is a salary increase of \$1,804.24 over the standard salary for direct physical restoration. The total salary figure for the site consists of the amount for the combined standard salaries shown in the preceding paragraph and the extra cost for consultation, \$25,879.83 + \$1,804.24, or \$27,684.07. At the other three sites, a test excavation is proposed to fully assess the magnitude of damage. This required the addition of two person days for fieldwork and one person day each for analysis and report preparation. The result is a salary increase of \$527.76 over the standard salary for direct physical restoration. The total salary figure for these sites consists of the amount for the combined standard salaries and the extra cost for the test excavation, \$25,879.83 + \$527.76, or \$26,407.59.

One other site is proposed for direct physical restoration salary above the standard level, but not for oil effect monitoring. Because this site has disinterred human remains, eight days of project supervisor time were added for consultation with Native Corporations at the

cost of \$1,804.24. Also, the large volume of disturbance at this site required the addition of four person days for fieldwork and two person days each for analysis and report preparation. The result is a salary increase of \$1,055.52 over the standard salary for direct physical restoration. The total salary figure for this site consists of the standard salary for direct physical restoration and the extra costs for consultation and restoration measures, \$3,155.79 + \$1,804.24 + \$1,055.52, or \$6,015.55.

Finally, two sites in the direct physical restoration only category have salary figures below the standard level because their injuries require measures different from those proposed for most other sites. For one site, the total salary figure of \$2,480.83 is for repatriation or reinterment administration and consultation. The direct physical restoration measures proposed for the other site allowed the elimination of two person days for fieldwork and one person day each for analysis and report preparation. The result is a salary decrease of \$892.08 from the standard salary for direct physical restoration to a total salary figure of \$2,263.71.

The other elements of the restoration cost proposals are support costs. Basic support costs are for: fieldwork per diem, transportation, supplies and equipment, and processing and duplication. Recovery of items requiring expenditures for curation is anticipated at all but two of the sites proposed for direct physical restoration measures. The proposals for three sites involve repatriation or reinterment costs. (At one site, the only support costs are for repatriation or reinterment.)

The support cost amounts vary by site. The average support cost figures are: for year one oil effect monitoring,

\$4,086.83; for years two through ten oil effect monitoring, \$33,159.60; and for direct physical restoration, \$10,920.33.

The total figures for the restoration cost proposals for the 24 sites under consideration are as follows:

Year one oil effect monitoring:	\$69,913.40
Years two through ten oil effect monitoring:	<u>\$529,791.30</u>
Ten years of oil effect monitoring:	\$599,704.70
Direct physical restoration measures:	<u>\$272,126.49</u>
Total restoration costs:	\$871,831.19

Average costs per site for oil effect monitoring and direct physical restoration were calculated from the total cost figures by dividing them by the number of sites for which the measures are proposed. The average costs are \$59,970.47 per site for oil effect monitoring and \$14,322.45 per site for direct physical restoration.

The projections for the numbers of sites injured on which the estimates of gross restoration costs are based were derived from four sets of figures:

1. The projected numbers of injured sites in the draft State University of New York report entitled *Exxon Valdez Oil Spill Archaeological Damage Assessment* by Albert J. Dekin et al. (1992).
2. The Dekin et al. figures reduced by the percentages corresponding to the number of sites eliminated from consideration by the damage assessment panel because the injuries were not severe enough to require restoration.
3. The number of sites proposed by the panel for oil effect monitoring and

direct physical restoration as percentages of the total number of sites now included in the Alaska Heritage Resource Survey site inventory records for the oil spill area.

4. The number of sites proposed by the panel for oil effect monitoring and direct physical restoration as percentages of the number of sites actually located and examined by Exxon crews during oil spill response activities.

Each of these four figures was multiplied by the average per site costs for oil effect monitoring and direct physical restoration. The resulting gross restoration costs estimates for oil effect monitoring are:

- (1) \$31,844,319.57
- (2) \$28,126,150.43
- (3) \$1,439,291.28
- (4) \$3,598,228.20

For direct physical restoration measures, they are:

- (1) \$4,840,988.10
- (2) \$3,867,061.50
- (3) \$658,832.70
- (4) \$1,618,436.85

The total gross restoration cost estimates are:

- (1) \$36,685,307.67
- (2) \$31,993,211.93
- (3) \$2,098,123.98
- (4) \$5,216,665.05

The gross cost estimates based on the number of sites actually field checked by Exxon (number four above) are seen as reliable indicators of the overall magnitude of archaeological restoration needs resulting from the oil spill.

Two important conclusions are drawn from the results of the work summarized in this paper. First, the ARPA damage assessment model was used successfully to generate credible restoration cost determinations for the documented archaeological injuries of the *Exxon Valdez* Oil Spill. Second, the ARPA model should be the damage assessment and restoration cost determination standard for archaeological injuries resulting from future oil spills or other similar situations.

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Long-Term Social Psychological Impacts of the Exxon Valdez Oil Spill

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During the last decade scientific studies have been conducted on the community impacts of technological disasters (Baum et al., 1982, 1983; Omohundro, 1982; Couch and Kroll-Smith, 1985; Gill and Picou, 1989). Most recently, a number of studies provide empirical evidence which demonstrates that, in contrast to natural disasters, technological disasters produce long-term patterns of stress. These patterns of stress appear to be related to issues of "uncertainty" of extent of contamination (Vyner, 1988), protracted litigation activities (Brown and Mikkelsen, 1989; Picou and Rosebrook, 1992), and general sociocultural disruption (Freudenberg and Jones, 1991).

Social impact assessments of the Exxon Valdez oil spill have been relatively limited. Studies suggest that subsistence activities of Alaskan natives were initially disrupted by the spill (Fall, 1990; Restoration Planning Work Group, 1990; Dyer et al., 1992). Furthermore, research also reveals the existence of social impacts some 18 months following the spill (Picou et al., 1992). The nature of these impacts included relatively high levels of family, work and personal disruption, as well as continuing patterns of personal stress (Picou et al., 1992).

Given this recent interest in identifying long-term social impacts of technological disasters and the paucity of longitudinal data on this subject, this study will present and evaluate a causal model which depicts long-term social psychological impacts.

A disaster impact assessment design structured the methodological procedures of this research (Gill and Picou, 1991; Picou and Rosebrook, 1992). This approach included standardized indicators of social impacts, random sampling procedures, and a control community comparison. The survey instrument included demographic, social and psychological indicators used in previous disaster research. This research design used stratified random sampling techniques and included personal, telephone and mail surveys over the three year data collection period.

Impact and Control Community Description

Cordova was selected as an impacted community because of its economic dependency on commercial fishing and its cultural heritage of subsistence activities. It is located in the southeastern region of Prince William Sound. The community is isolated from other communities by mountains, glaciers, rivers and the sea. No roads have connected Cordova to the outside world since 1964. Incorporated in 1909, the city became an export center for copper mined in the Wrangell mountains to the north. The closing of the copper mines in 1939 led to increasing involvement in commercial fishing. The community population currently fluctuates from 2,500 during the winter to over 4,500 during the summer commercial fishing season.

Cordova fishermen own almost one-half (44%) of all Prince William Sound

herring permits and 55% of all Prince William Sound salmon fishery permits (Stratton, 1989). Subsistence activities (i.e., harvesting, giving, receiving fish, moose, deer, berries, etc.) characterize 90% of Cordova's households (Stratton, 1989) and Alaskan natives make up 18 percent of the population. These and other data classify Cordova as a natural resource community (Dyer et al., 1992). The commercial fishing industry and numerous related businesses link this community directly to seasonal harvests of renewable natural resources.

Petersburg, Alaska was selected as the control community for this study. Petersburg is located on an island in the southeastern part of the state and is relatively isolated with no road connections outside Mitkof Island. Petersburg has a population of 3,500 people, with Alaskan natives comprising approximately 20%. Petersburg's economic base is based on commercial fishing. Petersburg residents also engage in subsistence activities at a rate similar to Cordova (Smythe, 1988; Stratton, 1989).

Data Collection: A stratified random household sample was selected in the Cordova community. In August, 1989, personal interviews with 86 respondents were conducted, reflecting 70 households. Random digit dialing telephone interviews were conducted in Petersburg and Cordova to complete data collection activities for the first year. In 1990, follow-up interviews were conducted by mail and telephone surveys. In 1991, respondents in Cordova were reinterviewed by personal interviews and respondents in Petersburg were once again contacted by telephone.

Indicators and Measures: Information was collected from respondents on demographics, social attitudes, work,

family and personal disruption and social psychological stress. Social psychological stress was measured by the "impact of events scale," which identifies two stress components—intrusion and avoidance (Horowitz et al., 1979; Horowitz, 1986).

Statistical Analysis: A series of path models, or structural equation models, were calculated for data available from 1989 to 1991 (Duncan, 1966; Birnbaum, 1981). In general, these models attempt to identify causal relationships between social structural characteristics, social disruption and psychological stress.

Results

Higher levels of intrusive stress and avoidance behavior were observed for the impacted community in 1989, 1990 and 1991. These differences were found to be statistically significant ($Pr < .05$) when t-tests were applied to compare impact and control community mean scores on the stress indicators. Mean stress scores were found to decline from 1989 to 1991 in the impact community suggesting that, over time, a reduction in the intensity of the social psychological impacts.

The evaluation of the models provided limited support for the hypothesis that social structural characteristics influence social and psychological reactions of victims of technological disasters. The primary predictors of intrusive stress in 1991 included intrusive stress in 1990 and 1989, work disruption in 1989 and continuing social disruption in 1991. Attitudes toward the effectiveness of cleanup operations were found to predict long-term stress. That is, respondents who were most pessimistic about cleanup effectiveness in 1990 tended to be more stressed in 1991. In general, respondents who were male and who

experienced both work and family disruption in 1989 held the most pessimistic views of cleanup effectiveness.

The long-term patterns of social psychological stress found in previous studies of a variety of technological disasters were also observed for residents of the impact community in this research. Higher levels of intrusive stress and avoidance behavior were found to exist in 1989, 1990 and 1991 in the impact community. Over time, levels of stress were found to be declining. For example, mean intrusive stress scores fell from 24.47 in 1989 to 19.32 in 1990 and then further declined to 17.74 in 1991. This general trend suggests a pattern of return to community equilibrium.

Attempts to develop and evaluate causal models of long-term stress resulting from the Exxon Valdez oil spill were modestly successful. Although some intervening variables were found to predict long-term stress, initial stress levels were the most important predictors of later stress levels in the models. Intrusive stress existing some 20 months after the spill was found to be related to perceptions that the cleanup was ineffective. However, the relative effects of both attitudes toward the cleanup and problems experienced from litigation on intrusive stress were small when compared to effects of previous stress levels generated from the spill.

In conclusion, alternatives to linear, additive models may be required to fully understand the complex patterns of long-term stress created by technological disasters in general and the Exxon Valdez oil spill, in particular. Models utilizing interaction terms may provide more accurate explanations. Future analyses of these data will evaluate the utility of this approach.

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The Economic Impacts of the *Exxon Valdez* Oil Spill on Southcentral Alaska's Commercial Fishing Industry

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The potential of natural disasters to generate short-term economic benefits for impacted individuals and communities has become an accepted social science notion (Dacy and Kunreuther, 1969; Rogers, 1970; Cochrane, 1975; Wright et al. 1979; Friesema, et al. 1979; Rossi et al. 1983). Unfortunately, the economic dimensions of human-made disasters have not received similar treatment despite the considerable attention that has recently been focused on these events and their sequelae (e.g., Erickson, 1976, 1991; Levine, 1982; Couch and Kroll-Smith, 1985; Shrivastava, 1987).

In the case of oil spills, economic research has been largely confined to the estimation of comprehensive damage assessments (Mead and Sorensen, 1970; Burrows et al, 1974a, 1974b; Brown et al. 1983; Grigalunas, 1982; Grigalunas et al. 1986; Assaf et al. 1986). With the exception of work completed by Nelson (1981) and Restrepo (1982) there has been little consideration of the perturbing influences that technological disruptions can have on local and regional economies.

Within the context of recent technological accidents (e.g., Bhopal, Three Mile Island, Love Canal), the *Exxon Valdez* oil spill is distinguished by its widespread physical damage of a highly valued natural environment and its extraordinary economic bonanza (Cohen, 1993). In order to contain the spilled cargo, collect contaminated debris, and clean oiled shorelines a massive emergency response operation was assembled that eventu-

ally employed 11,000 local residents and transient laborers at wages exceeding \$16 per hour.

However, while these ephemeral windfall profit opportunities were being exploited, the fundamental component of the regional economy was experiencing a downward realignment. In an effort to measure the magnitude of this adjustment, this analysis derives ex post estimates of the oil spill's economic impacts on southcentral Alaska's commercial fishing industry during the years 1989 and 1990 in isolation of the considerable financial benefits imparted by the emergency response operation. A three-phase methodology is employed to determine the ex-vessel revenue that would have been earned for each of southcentral Alaska's eleven major commercial fishery products (chinook, sockeye, coho, pink and chum salmon, king crab, tanner crab, Dungeness crab, Pacific herring sac roe, Pacific halibut, and sablefish) during each of these two years had the oil spill not occurred.

First, estimates of the accident's harvest volume impacts are constructed using data reported by the Alaska Department of Fish and Game (ADF&G), International Pacific Halibut Commission (IPHC), and the National Marine Fisheries Service (NMFS). The pre-season harvest expectations for each commercial fishery in southcentral Alaska (Prince William Sound, Lower Cook Inlet, and Kodiak Island) were compared to actual yields. In order to ascertain the extent of

contamination from the oil spill, physical monitoring and organoleptic test results were examined for each regulatory jurisdiction that evidenced a harvest deficit in either of the two years under consideration. Harvest volume impacts for each commercial fishery were then apportioned to the accident according to these data. This method led to the conclusion that the oil spill's harvest volume impacts were confined principally to Pacific herring and pink and chum salmon during the 1989 season. No harvest volume reductions attributable to the accident were experienced in 1990.

Second, the *Exxon Valdez* oil spill was hypothesized to have motivated a fundamental shift in ex-vessel demand for southcentral Alaska's commercial fishery products as retail consumers became fearful of tainted seafood. This effect was estimated with a price-dependent demand model that was variously adapted to the specific characteristics of each of the region's major fish and shellfish species. Independent variables included in these specifications were the seasonal quantity of landed product, national income of major consuming countries, frozen and canned inventory holdings, price of substitutes, and exchange rates between the United States and major consuming countries.

The structural equations derived from this model were estimated by a biased least squares fitting procedure with data for the period 1964-1988. The subsequently derived coefficients were then used to generate ex post forecasts of the ex-vessel prices that would have prevailed for southcentral Alaska's commercial fishery products during 1989 and 1990 in the absence of the oil spill. The predicted ex-vessel prices were then con-

trasted with their corresponding actual values. On the basis of this technique, the largest ex-vessel price impacts were sustained in both years by Pacific herring sac roe and coho and chum salmon. Noteworthy is the observation that actual ex-vessel prices for Pacific halibut and sablefish exceeded predicted values during both seasons, raising the possibility that some demand substitution of these products least threatened by oil contamination may have occurred.

Finally, the full extent of these estimated harvest volume and ex-vessel price impacts cannot be attributed to the *Exxon Valdez* oil spill as the commercial fishing industry was concurrently perturbed by several other biological and economic influences in addition to the accident. On the biological side, commercial fishery yields are generally subject to considerable stochastic variability due to numerous environmental factors. For instance, fluctuations in rates of predation, disease mortality, and water temperature can alter interseasonal commercial fishery harvest volumes. Additionally, various forms of human intervention, including regulatory measures and artificial cultivation, can result in harvest volume adjustments.

On the economic side, ex-vessel prices in 1989 and 1990 were influenced by several perturbations that occurred simultaneously to the oil spill. After attaining unprecedented levels in 1988, ex-vessel prices for most Alaskan commercial fishery products began to erode in 1989. This trend was motivated by substantial increases in the volume of salmon produced internationally. Other factors contributing to this decline included excessive wholesale inventories, reduced consumer spending, unfavorable exchange rate adjustments, and suspended

speculation among Tokyo fishery product wholesalers. In order to isolate the ex-vessel revenue impacts of the *Exxon Valdez* oil spill from these confounding biological and economic factors several alternative scenarios were simulated.

Though each scenario was based on different assumptions regarding the magnitude of the *Exxon Valdez* oil spill relative to the array of confounding factors, a consistent conclusion emerges. The economic boom that swept over southcentral Alaska as a result of the oil spill obscured the decline in the profitability of the region's commercial fishing industry and exasperated the deterioration of international market conditions. Specifically, the accident reduced ex-vessel revenue for southcentral Alaska's commercial fishers during 1989 by an amount ranging between \$6.4 million and \$41.8 million. Ex-vessel revenue reductions were greatest for the region's commercial harvest of sockeye and pink salmon while increased ex-vessel values for Pacific halibut and sablefish marginally mitigated these declines. This analysis indicates that the oil spill's ex-vessel revenue impacts in 1990 were between \$11.1 million and \$44.5 million.

Employing 1988 as a baseline, these amounts represent 3-19 percent of the ex-vessel value of southcentral Alaska's commercial fishing economy. Given the considerable imprecision inherent in economic impact analysis of complex perturbations such as the *Exxon Valdez* oil spill, more explicit evaluation is not readily possible. In spite of this indeterminacy, this analysis provides a bounded interval in which one measure of the accident's economic dimensions can be considered.

The interpretation of these results requires the consideration of at least four

factors. First, the harvest volume impact methodology lacks scientific rigor and is further undermined by the uncertainty that exists regarding hydrocarbon-based toxicity on individual marine species. Even under relatively static circumstances the determination of fishery harvest volumes on the basis of historical trends and biometric forecasting models is an imprecise exercise.

Second, the ex-vessel price model provides only a partial explanation of demand behavior and its empirical specification relies on commercial fishery product data not generally noted for a high degree of reliability.

Third, changes in the market characteristics of many of southcentral Alaska's fishery products over the course of the specification period raises concerns as to parameter stability in the ex-vessel price model.

Finally, many of southcentral Alaska's commercial fishers were employed in various capacities by the cleanup operation mobilized after the oil spill. Remuneration for boat rentals and services exceeded by several orders of magnitude the ex-vessel revenue lost due to the inability to conduct scheduled commercial fisheries, and overall economic performance must be viewed in this light.

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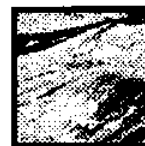
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Detecting Population Impacts from Oil Spills: A Comparison of Methodologies.

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The simplest response of a population to an oil spill is an immediate kill. A certain proportion of the population is killed by the spill, and then the population gradually recovers. A second possibility is that there is continued post-spill impact on the population due to reduced reproduction, growth or some other mortality, but this impact disappears over time. A third possibility is that there is some permanent impact on the population, so that it never would recover to its pre-spill abundance. This could be due to permanently lost habitat, change in the community composition, or permanent changes in survival rates.

There are four ways that the impacts on a population can be detected. These are (1) direct body counts of the number of animals killed, (2) pre vs. post spill comparison of population sizes, (3) oiled vs. unoiled spatial comparisons of abundance after a spill, and (4) direct measurement of vital rates in oiled vs. unoiled sites.

(1) Body counts are the number of individuals killed by the spill and can sometimes be directly measured, or estimated. Such counts are available for several bird and marine mammal species. However, body counts do not provide, by themselves, any evidence of population level impact. The body counts must be considered in relation to population abundance, natality and mortality rates. Thus any quantitative assessment of population level impacts will require some

form of population dynamics model, combined with several types of data. (2) When abundance surveys are available, comparison of pre and post spill numbers may be made to assess the change in population. The statistical power of such comparison will depend upon the reliability of the census method, the natural variability of the population, and the magnitude of change induced by the spill. This method can clearly not be used when no pre-spill abundance data are available, as was the case in many fish species. Pink salmon exhibit such high year-to-year variability that only the most severe of impacts could have been detected by this method. We show that the good returns of pink salmon to Prince William Sound in 1990 and 1991 do not provide strong evidence that there was no significant impact of the oil spill.

(3) When the populations can be spatially stratified into oiled and unoiled sites, it may be possible to assess the impact of the spill even when pre and post spill data are not available. The abundance of individuals in oiled sites is compared to the abundance in unoiled sites. Again the power of this method will depend on the reliability of the census method, the natural variability from site to site, and the magnitude of the change induced by the spill. A key problem posed by this approach is the fact that treatments (oiled vs. unoiled) were not

assigned randomly, and post spill differences may reflect underlying habitat differences rather than the impacts of oiling. The study of impacts on Dolly Varden and cutthroat trout illustrates how such oiled vs. unoiled comparisons can work (Hepler et al., 1993). This is contrasted to pink salmon studies where oiled vs. unoiled abundance data provide little statistical power (Sharr et al., 1993).

- (4) A final approach is to measure life history parameters in oiled and unoiled sites. The estimated parameters are used in a life history model to estimate population level impacts. The differences in growth observed in oiled vs. unoiled sites (Hepler et al., 1993), and in egg survival for pink salmon (Bue and Sharr, 1993) are examples of how this approach can provide evidence of damage even where population level damages may be difficult to measure directly.

The weakness of this approach is it depends on the validity of the population dynamics models used, and in most cases the extent of damage depends on the level of compensatory mortality in

the life history after the damage. If there is strong density dependent mortality then the population level impacts will be much less than the raw mortality caused by oiling. The potential for compensatory mortality significantly decreases the power of this approach.

A final issue we consider is the statistical framework for analysis of damage studies. An obvious question encountered in any study of oil spill impacts is what level of significance should be used. We argue that traditional hypothesis testing statistics should not be used, but rather the statistical product of an analysis should be a Bayesian posterior distribution on the intensity of oil spill impact. Hypothesis tests have little if any meaning in testing for impacts, and should be abandoned.

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Coastal Habitat Studies: The Effect of the Exxon Valdez Oil Spill on Shallow Subtidal Fishes in Prince William Sound.

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Fish communities were monitored in shallow (< 20 m) depths of paired oiled and unoiled (control) study sites within Prince William Sound in 1990, using diving transect techniques. Habitats studied were defined by dominant macrophytes (macroscopic plant), including eelgrass and *Agarum/Laminaria*.

There were four pairs of oiled and unoiled (control) eelgrass study sites. At each site, we established three 30 m long transects running parallel to shore, in approximately the middle of the depth range of eelgrass and at randomly selected locations along a 200 m section of shoreline selected for sampling. Two divers counted fishes along each transect within 1 m on either side and within 3 m of the bottom.

Agarum/Laminaria habitats were subdivided into bay and point categories, and further stratified into shallow (2 - 11 m) and deep (11 - 20 m) depth zones. There were three pairs each of bay and point oiled and control sites. In each depth zone three 30 m long transects were randomly located and run parallel to the shoreline. Fish were counted as in the eelgrass habitat.

The null hypotheses of no difference among oiled and control sites was tested with a blocked ANOVA, with oil category as the main effect and pair as a blocking factor. Homogeneity of variance was tested with Levene's test (Milliken and Johnson, 1984). If data failed to meet the assumption of homogeneity, and if the significance level for

the ANOVA was less than 0.10, we tested the significance of the ANOVA F statistic with a randomization procedure (Manly, 1991).

Over 15 species of fishes were observed in the eelgrass transects, although numbers were dominated by young of the year (YOY) Pacific cod (*Gadus macrocephalus*), which were the most abundant species in both oiled (0.84 of all fishes) and control sites (0.46 of all fishes). Other important taxa were gunnels (Pholidae), greenlings (Hexagrammidae), and Arctic shanny (*Stichaeus punctatus*). The number of all fishes per transect (60 m²) ranged from 5.3 to 105.0, with highest numbers in the oiled sites in each of the four site pairs. The abundance of all fishes, and of YOY Pacific cod, were significantly ($p < 0.01$) higher at oiled sites.

Over 30 species of fishes were observed in *Agarum/Laminaria* habitats, and the community was dominated by Arctic shanny and a mixed group of sculpins. For the purposes of our study we divided the sculpins into small and large species categories. In *Agarum/Laminaria* bays, Arctic shanny, most of which were YOY juveniles, were the most abundant species in shallow depths (0.63 of all fish in oiled sites and 0.31 in control sites) and were significantly more abundant at oiled sites. Total fish abundance was also higher at oiled sites as a result of the high numbers of Arctic shanny. In the deep depths of bays, small sculpins (mainly *Artedius* sp.) were the most abundant fishes. There were no differences in fish

abundances in the deep strata of bays.

At *Agarum/Laminaria* study sites on points, small sculpins were the most abundant fishes in shallow depths and were significantly more abundant at oiled sites. Greenlings were more abundant at control sites ($p < 0.05$). In the deep strata of the point habitats greenlings were significantly more abundant at oiled sites, as was the abundance of all fishes combined ($p = 0.03$).

In both eelgrass and *Agarum/Laminaria* habitats, oiled sites had overall higher abundances of fishes than did control sites due to the higher numbers of the dominant YOY fishes. Ebeling et al. (1972) also found greater numbers of larval and young of the year fishes in oiled sites relative to control sites in the Santa Barbara Channel after a large oil spill there in 1969. Higher abundances of YOY fishes may have resulted from greater settlement of pelagic larvae into oiled areas, from lowered mortality of YOY juveniles after settlement, or from post-settlement migration of juveniles into oiled sites or away from control sites. Our data do not provide a basis for distinguishing among these hypotheses. However, other components of the overall subtidal program measured the abundances of many invertebrates at the same locations where fishes were counted. Consequently, we conducted a feeding study of YOY Pacific cod from eelgrass study sites to determine if prey use pat-

terns were influenced by availability of invertebrate prey taxa.

We examined the feeding habits and condition (based on Fulton's length/weight index) of YOY Pacific cod from oiled and control eelgrass sites. Fish from oiled sites tended to have higher volumes of stomach contents, and had eaten proportionally more mollusk larvae than fish from control sites, where crustaceans were more common in diets. These differences in diet appear related to the relative availability of prey in oiled versus control eelgrass study sites. Fulton's condition index did not differ in any consistent pattern between oiled and control study sites. Our results suggest that in eelgrass habitat the increased abundance of YOY Pacific cod at oiled study sites is related to the increased diversity and abundance of epifaunal suspension-feeding prey taxa, especially mussels.

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Survey of Oil Exposure and Effects in Subtidal Fish Following the Exxon Valdez Oil Spill: 1989-1991

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Petroleum and its components have the potential to damage fishery resources. Of special concern in Prince William Sound and areas in the Gulf of Alaska affected by the Exxon Valdez oil spill, are species such as Dolly Varden char and adult salmon, which inhabit or pass through the littoral zones, and benthic fish species which live in subtidal areas in close association with bottom sediments. Numerous laboratory studies have demonstrated that exposure of fish to petroleum hydrocarbons can result in a variety of adverse effects.

However, because of the high rate of metabolism of petroleum hydrocarbons by many species of fish, direct measurement of tissue concentrations of parent compounds does not generally provide a useful indicator of exposure of fish to petroleum hydrocarbons. Therefore, it has been difficult to document exposure of fish to petroleum following oil spills.

In recent years, however, methods have been developed for determining such exposure, based on our knowledge that many of the metabolites of aromatic petroleum hydrocarbons are fluorescent, and a primary route of excretion of these metabolites is through the bile. Thus we have demonstrated that the measurement of fluorescent aromatic compounds (FACs) in fish bile serves as a useful indicator of petroleum exposure in field-sampled animals (Krahn et al., 1992; Krahn et al., this volume). Such measurements are now a mainstay of many as-

essment and monitoring programs, including those following the Exxon Valdez oil spill. Additionally, it is known that certain forms of cytochrome P450 can increase dramatically following exposure to a variety of exogenous compounds. Of most interest for aquatic monitoring programs is the finding that an increase in hepatic cytochrome P4501A (P4501A) appears to be a useful indicator of exposure of fish species to a wide variety of organic contaminants, including many compounds contained in petroleum (Payne et al., 1986; Collier and Varanasi, 1991).

In this paper we present summaries of our assessments of exposure of several species of fish for three years after the Exxon Valdez oil spill, using these two methods (biliary FACs and hepatic P4501A). Data are presented for Dolly Varden char (*Salvelinus malma*), yellowfin sole (*Limanda aspera*), rock sole (*Lepidopsetta bilineata*), flathead sole (*Hippoglossoides elassodon*), Pacific halibut (*Hippoglossus stenolepis*), and pollock (*Theragra chalcogramma*).

Our studies of exposure of adult Pacific salmon species are presented by Stein et al. in this volume. In addition to assessing petroleum exposure, samples were also taken for examination of histological alterations and assessment of reproductive function in two species, Dolly Varden and yellowfin sole, because of the potential for petroleum and related compounds to adversely affect tissue

structure and reproduction. The data reported here are in units of ng phenanthrene equivalents/mg bile protein, or ng/mg, for levels of FACs in bile, and pmoles benzo[a]pyrene metabolized/mg microsomal protein•minute, or pmol/mg, for P4501A-associated enzyme activity. Methodology used for obtaining these results can be found in the detailed study plans prepared for the Natural Resources Damage Assessment Trustee Council. More detailed presentations and analyses of the data are being prepared for publication in peer-reviewed scientific journals.

Summaries of findings by species

Dolly Varden char, sampled in the littoral zone by either beach seines or gill nets run perpendicular to the shore, showed some of the highest levels of exposure to oil (as measured by levels of FACs in bile, up to mean values of $\approx 15,000$ ng/mg) of any fish sampled in 1989. However, by 1990, these levels had dropped markedly (to $< 5,000$ ng/mg) at heavily oiled sites such as Tonsina Bay and Snug Harbor. Hepatic P4501A activities were also elevated (> 200 pmol/mg/min) in Dolly Varden at these oiled sites in 1989, and had dropped by 1990 (< 100 pmol/mg/min). These results suggest that the heavy oiling of the intertidal area seen in 1989 affected fish in the very nearshore subtidal area, such as Dolly Varden char, and by 1990 the levels of petroleum hydrocarbon contamination of these areas were substantially reduced. In 1990, Dolly Varden tissues were also analyzed histologically, but no increases in prevalences of histopathological conditions of liver, kidney, gonad, or gill were seen in conjunction with apparent oil exposure. Additionally in 1990, measurements of plasma estradiol concentrations and ovarian maturation

in female Dolly Varden captured at 12 sites showed little evidence of reproductive impairment in this species, although plasma estradiol levels tended to be lower in the animals most heavily exposed to petroleum. Because there was a substantial decrease in petroleum exposure between 1989 and 1990, Dolly Varden were not sampled in 1991.

Exposure to petroleum was readily discernible in the benthic flatfish species, yellowfin sole, though the levels of FACs in bile (up to $\approx 10,000$ ng/mg) were less than for Dolly Varden, suggesting less exposure to oil. However, levels did not drop markedly between 1989 and 1990 at oiled sites. By 1991, however, substantially decreased exposure was evident at Snug Harbor. In both 1990 and 1991 there was evidence of increased P450 activities in yellowfin sole at oiled sites. Similar to Dolly Varden, there was little evidence of reproductive dysfunction in the female yellowfin sole from the oiled sites in 1990, although there was again a trend toward lower levels of plasma estradiol in the most heavily exposed fish.

Levels of biliary FACs and hepatic P4501A activities were determined in rock sole from several sites in 1989, 1990, and 1991, and rock sole were also examined for the presence of histopathological lesions in 1990. Similar to yellowfin sole, there was evidence of exposure to petroleum in rock sole sampled near oiled sites in 1989 and 1990, again up to $\approx 10,000$ ng/mg. Decreased exposure was observed at oiled sites in the limited sampling done in 1991. The results of histological analyses of tissues collected from this benthic species in 1990 showed no alterations in liver, kidney, or gonad histology. However, there was a significantly ($p < 0.005$) increased prevalence of respiratory epithelial hyperplasia (REH)

of the gill at three sites where the biliary FAC data suggested that oil exposure was greatest (Tonsina Bay, Snug Harbor, and Sleepy Bay) as compared to the prevalence in fish collected from Olsen Bay and Rocky Bay. Additionally, the severity of gill REH was significantly greater ($p < 0.05$) at the three more impacted sites.

Increased levels of biliary FACs and induced hepatic P4501A activities were measured in flathead sole from heavily oiled sites in 1989 (FACs only) and 1990 (FACs and P4501A). In 1991 these same measures were slightly elevated in sole from Snug Harbor compared to a less impacted site, Olsen Bay. These results are similar to observations in yellowfin sole and rock sole, which is consistent with all three of these flatfish species being captured from similar habitats. No histological analyses of tissues from flathead sole have been done.

Pacific halibut, captured generally at depths of >30 m, showed some evidence of increased oil exposure in 1989, as determined by levels of FACs in bile, but levels were substantially less (<7000 ng/mg) than for other flatfish species captured at shallower depths. By 1990 these levels of FACs had dropped considerably (to <2500 ng/mg) at Tonsina Bay and Snug Harbor.

Pollock were not sampled until the late winter of 1990, and then were only sampled for levels of biliary FACs. At that time increased levels of FACs (up to ≈ 8000 ng/mg) were evident, especially in pollock from inside Prince William Sound, and by 1991 these levels had dropped quite substantially, though it was still possible to detect increased levels at some sites inside PWS, compared to animals collected from unimpacted or distant sites. Assessment of reproductive function in female pollock collected

in 1991 did not show any substantial effects that could be positively ascribed to increased oil exposure.

The results of analyses of oil exposure in several species of subtidal fish following the *Exxon Valdez* oil spill definitively point to the necessity of monitoring the subtidal environment following major oil spills. The littoral zone appeared to be heavily impacted in the months immediately following the spill, but a year later, after a winter storm cycle, exposure of fish inhabiting this zone (i.e. Dolly Varden) was substantially decreased. In contrast, nearshore benthic fish species (up to ~ 30 m depth, species sampled were rock sole, yellowfin sole, and flathead sole) showed continuing exposure through the first two field seasons after the spill, and even after more than two years there was still some evidence of increased exposure of fish from these habitats.

Beyond this depth (>30 m), the degree of exposure of Pacific halibut appeared to be less than in benthic fish residing at shallower depths. A surprising finding was the evidence of exposure of pollock to petroleum approximately one year after the spill, at a site (e.g. Tugidak Island) more than 400 miles from the grounding site. Pollock, a major fisheries resource in Alaskan waters, are bathypelagic fish which feed in the water column. Thus these results suggest that the spilled oil affected either the water column or food supply of these fish at great distances from the spill, and for some time after the spill.

What remains to be determined are the potential impacts on fishery resources of long-term exposure to petroleum, albeit at moderate to low levels. To date, our studies have not shown any profound effects in the species studied following

the *Exxon Valdez* oil spill, but this finding is tempered by the delay in initiating studies of serious effects such as reproductive function, and the relatively short time over which such analyses have been conducted.

Although it will always be difficult to sample subtidal fishery resources comprehensively and rapidly following an oil spill, a better understanding of the potential impacts of petroleum exposure on fishery resources can be obtained through careful and realistic laboratory exposure of fish to petroleum. Such studies will need to go beyond the relatively straightforward short-term exposure studies which have been commonly done in the past. However, recent advances in methodology for assessing oil exposure in fish, together with current knowledge

of the biological processes involved in reproduction, immune function, and growth and survival of fish species, make this an appropriate course of action.

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Impact of Oil Spilled from the *Exxon Valdez* on Survival and Growth of Dolly Varden and Cutthroat Trout in Prince William Sound, Alaska

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From 1989 through 1991, the impact of spilled crude oil from the *Exxon Valdez* on growth and survival of anadromous Dolly Varden (*Salvelinus malma*) and cutthroat trout (*Oncorhynchus clarki*) in Prince William Sound were studied in accordance with the Clean Water Act and the Comprehensive Environmental Response, Compensation, and Liability Act. At the time of the spill, anadromous Dolly Varden and cutthroat trout were in lakes around Prince William Sound. Past studies on the behavior of these fish have shown that Dolly Varden and cutthroat trout spend their winters in lakes, emigrate to the sea during late spring, and return to the same lakes the following fall. Survival and growth rates in emigrating populations could be calculated from fish recaptured during the next spring emigration.

Five such populations of each species were intercepted with weirs in 1989, 1990, and 1991 during their annual seaward emigration to Prince William Sound in the spring. Two populations emigrated into the wake of the spill, while three emigrated into waters free of spilled crude oil. Our study populations were comprised of tagged adults and subadults of both species. Growth was measured directly on recaptured fish. Survival rates were estimated with log-linear models of capture histories of tagged fish.

A two-stage simulation based on bootstrapping and Monte Carlo techniques was used to compare average survival rates between study populations

that were and were not associated with spilled oil.

Ten thousand eight hundred fifty seven Dolly Varden and 1,086 cutthroat trout were tagged and released in 1989; and 41,510 and 2,496, respectively, in 1990; 46,286 Dolly Varden and 2,701 cutthroat trout were inspected for tags in 1990; and 28,657 Dolly Varden and 5,062 cutthroat trout were inspected in 1991. Fewer fish were intercepted in 1989 because weirs were installed after the larger, older fish had emigrated.

Three hundred thirty-six fish were recaptured and reported by persons working or recreating around Prince William Sound during the summers of 1989 and 1990; few of these fish (18 Dolly Varden and no cutthroat trout) had been caught across the boundaries of the spill. Ninety to 100% of recaptured emigrants were recaptured at the same weirs in 1990 and 1991 at which they were released a year earlier. Of those fish that had strayed (298), few Dolly Varden and no cutthroat trout had moved across boundaries marking the extent of spilled crude oil. On average an unexpected 47% of survivors of both species evaded recapture in 1990 even though three of the five weirs were fish-tight that year. This evasion was incorporated into the log-linear models used to estimate survival rates.

Growth and survival rates were significantly lower in study populations associated with spilled oil. Growth from 1989-1990 was on average less in study

populations that emigrated into the wake of the spill: 24% and 22% less for recaptured subadult and adult Dolly Varden and 36% and 43% less for subadult and adult cutthroat trout.

This difference persisted through 1991 for cutthroat trout, but not for Dolly Varden; growth during 1990-91 of recaptured Dolly Varden in populations not associated with oil slowed. Averages of estimated survival rates from 1989 to 1990 were less in study populations associated with spilled oil: 36% and 40% less for subadult and adult Dolly Varden and 28% less for adult cutthroat trout. Bioaccumulation of petrogenic hydrocarbons in the food chain or chronic starvation from its collapse were hypothesized as the pathways that spilled crude oil had slowed growth and accelerated mortality of Dolly Varden and cutthroat trout. Other studies in Prince William Sound in the wake of the spill have shown a collapse in some of the populations of inter-tidal and subtidal invertebrates that are common prey for both Dolly Varden and cutthroat trout.

Our results are consistent with the occurrence of a deleterious impact on growth and survival of those Dolly Varden and cutthroat trout emigrating into the wake of the oil spilled from the *Exxon Valdez*. Because our results come from observation and not from experiment, our study can not confirm that impact. No information on growth and survival rates of fish in these populations was available prior to our study, nor could we control which populations were exposed to spilled oil.

However, preponderance of evidence, and not confirmation, is the usual rule of judgment for assessing impacts in accordance with the Clean Water Act and with the Comprehensive Environmental Response, Compensation, and Liability Act. Survival and growth rates in those study populations of Dolly Varden and cutthroat trout not associated with spilled oil in Prince William Sound were typical of published rates for these species elsewhere, while rates for fish associated with spilled oil were atypically lower.

Assessment of Damage to Demersal Rockfish in Prince William Sound Following the Exxon Valdez Oil Spill

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Populations of demersal rockfish in Prince William Sound were studied from 1989 through 1991 to assess potential injury due to the Exxon Valdez oil spill. Injury was evaluated by determining the concentrations of hydrocarbons and histopathological alterations in rockfish that inhabit reefs located in oiled and unoiled sites.

Oil spilled from the Exxon Valdez was the probable cause of death for demersal rockfish killed in Prince William Sound immediately after the spill. Approximately 20 dead rockfish were brought to the collection centers in Valdez and Cordova from sites of reported fish kills. Five of these rockfish were necropsied and it was concluded that crude oil was the probable cause of death of all five.

These results prompted additional testing for hydrocarbons in living rockfish. The 1989 hydrocarbon analyses showed that at least 11 fish of 30 rockfish tested from treatment sites had been exposed to oil within the two weeks prior to collection, while none of the 13 fish in control sites were exposed to oil. These two pieces of information prompted the 1990 and 1991 studies to sample and test rockfish for continued exposure to hydrocarbons.

Tissues were collected from rockfish for analysis at four sites (two oiled and two control) in Prince William Sound in both 1990 and 1991. In 1990, four sites were sampled along the outer Kenai Peninsula. Tissues from several species of demersal rockfish: yelloweye (*Sebastes*

ruberrimus), quillback (*S. maliger*), and copper (*S. caurinus*), were collected for histopathological evaluation. Rockfish tissues, as well as stomach contents, unconsolidated benthic sediments, and sessile suspension feeders were collected at each study site for analysis of hydrocarbons.

The proportion of samples from oiled sites with aromatic hydrocarbons and their metabolites in the bile was compared to the proportion of samples from control sites with contaminated bile. Evidence of exposure to hydrocarbons was indicated by elevated concentrations of phenanthrene and naphthalene-equivalent compounds in the bile in concert with chromatographic patterns characteristic of hydrocarbon contamination. Results indicate a significantly higher incidence of hydrocarbons in the bile of fish from oiled areas than control areas in 1989 ($P=0.005$), however there were no differences in 1990 ($P=0.9332$) or 1991 ($P=.8438$).

In 1990, nine histopathologic lesions were scored by pathologists. However only four—liver lipidosis, liver sinusoidal fibrosis, liver, and kidney macrophage aggregates—were compared statistically. These lesions were selected because they were the most likely to be caused by exposure to toxins. In 1991, 26 different lesions were scored by the pathologists, however only the same four that were tested in 1990 were tested in 1991. Results from the statistical analysis of these lesion scores from rockfish in

Prince William Sound in 1990 and 1991 indicated that rockfish were exposed to toxic agents. There were differences between control and treatment sites in Prince William Sound in two of the four liver lesion scores (liver lipidosis [$P=0.0016$] and liver sinusoidal fibrosis [$P=0.0118$]) in 1990 and one of four (liver lipidosis [$P=0.008$]) in 1991. No differences in lesion scores were seen between sites on the outer Kenai Peninsula in 1990.

The histopathologic evaluation was conducted blind, that is, pathologists did

not know if the tissues were from fish from oiled or unoiled sites. Upon completion of the histopathologic examination the pathologists "predicted" which sites were oiled based on qualitative analysis of lesion scores. The speculated exposure history was accurate for all four sites in Prince William Sound.

Subsequent statistical analysis using Principal Component Analysis was able to determine differences in oiled and unoiled sites in both 1990 and 1991. Differences were more significant in 1991 than in 1990 using this analysis.

Histopathologic Analysis of Chronic Effects of the Exxon Valdez Oil Spill on Alaska Fisheries

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To determine the long-term effects of the Exxon Valdez oil spill on fish resources in Prince William Sound, Alaska, sampling was begun in 1990, more than 12 months after the spill, on four types of fish: (1) Dolly Varden char (*Salvelinus malma*) adults; (2) Pacific herring (*Clupea harengus*) larvae and adults; (3) several rockfish species (*Sebastes* spp.); and (4) pink salmon (*Oncorhynchus gorbuscha*) larvae and adults. Organs chosen for histopathologic examination were those most likely to contain chronic or residual rather than acute lesions. Whole larvae, or adult liver, trunk kidney, spleen, and olfactory nares were sampled and preserved in Bouin's or 10% neutral buffered formalin solution.

Fixed tissues shipped to the Aquatic Toxicology Program, University of California, Davis, were labeled by site of origin (e.g., Rocky Bay), but not by type of exposure history (i.e., oiled vs. clean/control). To eliminate bias during histopathologic examination, tissues from each fish were assigned a random number used only to identify a given study but not a site of origin. Tissues were embedded in paraffin, sectioned at 5 μ m, and read in ascending numerical order. All lesions were semiquantitatively ranked (none = 0, mild = 1, moderate = 2, and severe = 3). After lesions were scored, significant lesions were identified using principal components analysis (PCA), and analysis of variance (ANOVA) of scale values derived from PCA was used to identify site differ-

ences. After histopathologic results were reported, actual exposure history was revealed. Multivariate ANOVA was used to differentiate oiled from clean sites, and also to account for other variables such as age or sex. Sampling was repeated in 1991 only if statistically significant differences between fish from oiled vs. clean sites were demonstrated in 1990.

Dolly Varden char in Prince William Sound congregate over winter (November–May) in a few freshwater lakes, but then split up and return to the mouths of their home streams to spawn and feed during the rest of the year (Andy Hoffman, personal communication). Some of these stream outlets were heavily impacted by the Exxon Valdez oil spill. In June 1990, livers were sampled from 12 Dolly Varden from each of five sites; in October 1990, liver, spleen, kidney, and olfactory nares were sampled from 14 to 20 fish from four of the five sites sampled in June. For the June samples, the first principal component explained 34% of the variability in the lesion scores, giving significant weight to hepatic lipidosis and hepatocellular megalocytosis.

Hepatic lipidosis is a common response associated with exposure of fish to a variety of different agents (Meyers et al., 1985) including petroleum hydrocarbons (Fletcher et al., 1979; McCain et al., 1978); however, other studies have shown a decrease in hepatic lipid stores in fish from oiled sites (Haensly et al., 1982). Megalocytosis is a marker of hepatocyte damage from a variety of insults (Hinton

et al., 1992), but has not previously been associated with exposure to oil. Scores from the two oiled sites were significantly different from three clean sites (ANOVA, $P = 0.001$), and were ranked in decreasing order for fish from Eshamy Creek weir (oiled, score = 1.7), Green Island weir (oiled, score = -0.17), Rocky Bay weir (clean, score = -0.43), Makarka Creek (clean, score = -0.47), to Boswell Bay (clean, score = -0.54). Only scores from Eshamy Creek weir were significantly different from other scores with Tukey's Studentized range test. For the October samples, the first principal component explained 16% of the variability, and oiled vs. clean differences were still highly significant ($P = 0.0001$).

Herring in Prince William Sound concentrate into a few select near-shore areas and spawn in April. Eggs hatch and movement patterns of resultant larvae are mostly unknown until they appear in bait fish industry catches approximately 18 months after hatch (Evelyn Biggs, personal communication). Movement of adult fish between annual spawns and potential for mixing of stocks are also unknown. Three types of histopathologic studies were completed with herring.

First, naturally spawned eggs were collected from oiled and clean sites, brought to the laboratory to hatch, and resultant hatchlings were examined for lesions (1989, 309 larvae; 1990, 189 larvae). Several of the larvae sampled in 1989 had cranial masses that resembled tumors; however, histopathologic examination revealed these to be a result of autolysis and not neoplasia. Vacuolation of the lens, a common finding in 1989 (both in oiled and clean larvae) but not 1990 larvae, was also a result of post-mortem autolysis. None of the herring larvae had significant histologic lesions.

In the second group of herring studies, tissues from adults were sampled in April 1989 (ten fish each from two oiled and two clean sites), Fall 1990 (50 fish each from one oiled and one clean site), and April 1991 (20 fish each from one oiled and two clean sites). In 1989, severe hepatic necrosis in 20% of herring from the two oiled sites (Naked Island and Rocky Point) clearly differentiated these fish from control fish (none of the fish from control sites had severe hepatic necrosis).

Interestingly, PCA failed to demonstrate significant differences based on exposure history of the herring sampled in 1989. Hepatocellular necrosis was described in wild fish sampled nine months after the 1978 Amoco Cadiz oil spill (Haensly et al., 1982). In analysis of lesion scores in 1990 samples, the first principal component explained 18% of the variability, and placed substantial weight on liver glycogen, liver macrophages, liver single cell necrosis, splenic macrophages, and kidney macrophages. Hepatocellular megalocytosis, though present, did not contribute to site variability. Macrophage aggregates, including those in liver, spleen, and kidney, have been used as indicators of contaminant exposure and more often as a generalized nonspecific response to several stressful stimuli (e.g., starvation, heat stress) in many studies (Blazer et al., 1987; Herraes et al., 1986; Wolke et al., 1985) and numbers of hepatic macrophage aggregates were increased in fish exposed to oil (Haensly et al., 1982).

Studies have shown that macrophage aggregates increase with age in healthy fish but are independent of age in stressed fish (Blazer et al., 1987; Hinton et al., 1992). Age of herring from the oiled site (Green Island, 5.6 ± 0.16 years)

was significantly greater than the clean site (Knowles Head, 2.4 ± 0.13 years). Multiple analysis of variance (MANOVA) revealed that site differences were highly significant ($P = 0.01$), whereas age differences did not contribute significantly to lesion scores ($P = 0.29$). Differences associated with potential oil exposure were no longer detectable in herring sampled in the spring of 1991. In the third study, adult herring were exposed to crude oil in the water (20 or 60% water soluble fraction) or per os (force-fed in gelatin capsules) in the laboratory (220 exposed fish, 20 controls) for up to ten days. Exposed fish developed hepatic necrosis, providing strong evidence that oil was the cause of hepatic necrosis in wild-caught herring in 1989, and supporting evidence that macrophage aggregates in herring captured in 1990 resulted from oil exposure.

Pink salmon in Prince William Sound spawn in streams during late summer. Eggs hatch in December and fry emerge and move out to sea in the spring. Young adults move out of Prince William Sound and mix with other stocks for much of their life, and mature adults return to spawn in the same stream their parents spawned in two years earlier (Sam Sharr, personal communication).

Larvae and juveniles were sampled and examined for lesions in 1989 (726 pre-emergent larvae from 23 sites), 1990 (158 fish from 16 sites), and 1991 (160 fish from 11 sites). Mature adults were examined for lesions in two separate studies in 1990: 20 males and 20 females from each of eight sites in study #1 (320 fish) and from each of five sites in study #2 (200 fish). PCA detected some significant differences among lesion scores of oiled vs. clean groups, but the scores (e.g., autolysis) that accounted for these dif-

ferences could not be attributed to oil exposure.

Rockfish in Prince William Sound establish home ranges and tend to stay on the same rock bed for years (Andy Hoffman, personal communication). Therefore, they are potentially an excellent group of fish to use for monitoring effects of perturbations such as the Exxon Valdez oil spill. In 1990, 121 rockfish (8 different species) were collected from four sites in Prince William Sound and four sites off the Kenai Peninsula. Nine lesions in the liver, spleen, and kidney were scored and analyzed using PCA; an orthogonal transformation was done to put more weight on macrophage scores in Factor 1. When species differences were ignored in ANOVA of Factor 1, oiled vs. clean differences were not significant ($P = 0.21$); however, when species differences were included, oiled vs. clean differences were significant ($P = 0.035$).

In 1991, 107 rockfish (either quillback, yelloweye, or copper rockfish) were collected from four sites in Prince William Sound. Twenty-six lesions in the liver, spleen, kidney, gill, and heart were scored; analysis was the same as for 1990 rockfish. When species differences were ignored in ANOVA, oiled vs. clean differences were not significant for Factor 1 ($P = 0.09$) but were significant for Factor 3 ($P = 0.001$); when species differences were included, oiled vs. clean differences were significant for Factor 1 ($P = 0.001$) but not for Factor 3 ($P = 0.27$).

Multivariate analysis of variance on the 1991 data showed significant oiled vs. clean differences whether species differences were ignored ($P = 0.005$) or considered ($P = 0.02$). Lesions contributing most to variability in 1991 included hepatic megalocytosis and

macrophage aggregates, and splenic macrophage aggregates. Rockfish were collected in 1991 nearly 2.5 years after the Exxon Valdez oil spill, but histopathologic analysis indicates that lesion differences were more significant in 1991 than in 1990. Therefore, additional sampling is proposed for 1993 to determine if toxic effects continue.

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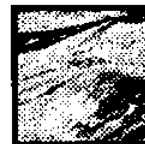
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Effects of the Exxon Valdez Oil Spill on Herring Embryos and Larvae: Sublethal Assessments, 1989-1991

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Embryo/larval development of Prince William Sound herring was evaluated from 1989 to 1991 to determine possible adverse effects from the *Exxon Valdez* oil spill. Herring embryos were collected from three replicate sites within each oiled or unoled location. Because herring did not spawn at the same sites every year, sites were generally different for each year.

Fairmont Bay, an unoled location, was sampled every year as was Rocky Bay, an oiled location on Montague Island. Naked Island, an oiled location, was sampled only in 1989 and 1990. Eggs were transported to the laboratory and incubated to hatch. Larvae were assessed for two types of sublethal damage, morphologic malformations and genetic effects. Relationships between the sublethal endpoints and hydrocarbon measurements from caged mussels were examined. Hydrocarbon measurements were ranked using principal components (PCA) specific to *Exxon Valdez* crude oil.

In 1989, larval malformations were significantly ($p < 0.05$) more severe at the two oiled locations (Rocky Bay and Naked Island) than at Fairmont Bay. Malformation scores at Naked Island were 55% higher than those from Fairmont Bay; differences were significant at all three depths. At Rocky Bay, scores were elevated by a mean of 37% but only the middle and high depths were significantly higher. The observed malformations included skeletal curvatures, cran-

iofacial defects, reduced cephalic differentiation, and finfold reductions. Larval defects are a standard manifestation of embryonic stress, whether from a toxic event or extreme environmental conditions such as high temperature (von Westernhagen, 1988; Weis and Weis, 1989).

Both morphologic endpoints, the severity of the malformations and the percentage of malformed larvae, ranked consistently with the level of oiling in 1989 by location. At Fairmont Bay, both endpoints showed the least damage and the mean PCA value was 2.0 (no to low oiling). Rocky Bay had intermediate values for sublethal damage and its mean PCA score (6.7, no to high oiling). Naked Island had the greatest sublethal damage and the highest mean PCA value (9.8, low to high oiling). The chemistry results reflected variability in the oiling levels within areas. Neither of the morphologic endpoints was significantly correlated with site-specific PCA scores.

Mitotic configurations in the pectoral fins were enumerated and evaluated for evidence of chromosome breakage (anaphase aberrations). Fairmont Bay larvae averaged 8.5 mitoses per fin and 10.4% aberrant anaphase configurations, values within normal limits. Larvae from the two oiled locations had reduced cell division and elevated anaphase aberration rates relative to Fairmont Bay larvae. At Rocky Bay, the aberration rate was 33.9%, 2.3 times higher than at the

unoiled site; at Naked Island, the aberration rate was 45.7%, 3.4 times higher. More individuals from the oiled locations thus exhibited genetic damage capable of reducing their subsequent survival. Reduced cell division and chromosome breakage result from exposure to genotoxic agents, including petroleum hydrocarbons (Longwell and Hughes, 1980).

Similar cytogenetic effects have been previously documented in fish eggs following oil spills (Longwell, 1977). All three cytogenetic endpoints (number of mitoses, % aberrations and % cytogenetically abnormal larvae) were ranked consistent with PCA scores among the three locations. Fairmont Bay larvae exhibited the least cytogenetic damage, Rocky Bay larvae were intermediate, and Naked Island larvae were most affected. Skeletal malformations and the anaphase aberration rate were significantly correlated with site-specific PCA scores. Skeletal and finfold defects, the anaphase aberration rate and the percentage of cytogenetically abnormal larvae were correlated with concentrations of aromatic hydrocarbons.

Subsequent sampling in 1990 and 1991 demonstrated that sublethal damage was undetectable. The extent of sublethal damage in Prince William Sound herring larvae declined from 1989 to 1990. The reduction in morphologic damage resulted in part from the improved incubation technique in 1990 but cytogenetic measurements were unaffected by the change. In 1990, morphologic endpoints were within baseline levels at all three locations but both the severity of malformations and the percentage of malformed larvae were significantly higher at Naked Island than at Fairmont Bay. Cytogenetic endpoints were within baseline

levels at Fairmont Bay and Naked Island, consistent with the decrease in hydrocarbon measurements at all sites evaluated in 1990. The mean PCA scores for Fairmont Bay and Naked Island were less than zero and that of Rocky Bay was 0.6. At Rocky Bay, the aberration rate and the percentage of cytogenetically abnormal larvae were significantly higher compared to the unoiled site. Because the 1990 hydrocarbon levels were also low at Rocky Bay, this residual genotoxicity might reflect sustained damage to the adults and thus be unrelated to mussel uptake measurements. To resolve this complicated picture, sublethal effects were more intensively studied the next year.

In 1991, morphologic and cytogenetic measurements were similar between Fairmont Bay and Rocky Bay. All endpoints were slightly higher than in 1990, probably reflecting the colder 1991 water temperatures. Whereas the aberration rate and percentage of cytogenetically abnormal larvae were significantly elevated at Rocky Bay in 1990, 1991 values were virtually identical to those from Fairmont Bay. Aberration rates were both 21.5%, slightly above the upper normal limit of 20%. At Fairmont Bay, 58% of the larvae were cytogenetically abnormal compared to 57% at Rocky Bay. These data suggest that oil-related toxicity was ameliorated by 1991 at Rocky Bay.

Herring did not spawn on Naked Island in 1991, so embryos were placed at three of the 1989 sites and allowed to develop *in situ*. Morphologic and cytogenetic analyses detected significant toxicity present at only one site which had been moderately oiled in 1989. Two 1989 sites at Rocky Bay were similarly evaluated, and consistently adverse sublethal

effects were not found. To summarize, These data suggest that the *Exxon Valdez* oil spill transiently elevated the incidence of sublethal damage in larval herring within certain affected areas but significant damage did not persist into 1991.

In the laboratory, herring embryos exposed to low concentrations of Prudhoe Bay water soluble fraction (WSF) exhibited sublethal damage similar to that observed in the field. The morphologic endpoints were significant at 0.10 to 0.48 mg/L WSF and the cytogenetic endpoints significant at 0.01 to 0.24 mg/L WSF. The anaphase aberration rate proved to be the most highly sensitive endpoint with rates elevated above the control at the lowest dose tested, 0.01 mg/L WSF. Significant differences at 0.10 mg/L WSF were detected with three other endpoints: craniofacial malformation severity, finfold malformation severity, and the percentage of cytogenetically abnormal larvae. All endpoints were significantly correlated with the log WSF dose. This presentation will also discuss relationships between the sublethal effects observed in the field and in the laboratory and measurements of *Exxon Valdez* crude

oil throughout Prince William Sound.

Sublethal impacts are an important part of toxicity assessments since they can be used to predict long-term damage or estimate effects on life stages that are difficult to study. For instance, sublethal effect data from herring can be integrated with measurements of egg abundance, embryo survival and larval densities to obtain estimates of embryo/larval success. Such estimates are essential to define potential toxic effects on fish populations.

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Herring Embryo Stage Sensitivity to Water Soluble Fraction of Prudhoe Bay Crude Oil

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The purpose of this study was: (1) To determine if short-term exposure to water soluble fraction (WSF) adversely affected developing embryos to different degrees depending on their developmental stage; (2) To establish an EC_{50} for Prudhoe Bay water soluble fraction following continuous embryo exposure for the entire incubation period; and (3) To evaluate previously oiled and unoled sites *in situ* for embryo toxicity.

To minimize inter-female variability, eggs from 8-10 females were randomly distributed to 100 slides at a density of 40-60 eggs/slide. Sperm from three to four males was then pooled and used to fertilize the eggs. After one hour, several slides were examined to verify fertilization success. At least 90% of the eggs had to be fertile in order to consider the spawning successful.

Slides containing fertile eggs were placed into a Plexiglas carrier to prevent slide-to-slide contact during transit, then submerged in seawater, gassed with O_2 and placed into a cooler containing wet ice for transport back to the University of Washington by commercial airliner. The total elapsed time from fertilization in Prince William Sound to arrival in Seattle was eight hours. Once at the University, the embryos were incubated at 8°C in 29 ppt seawater in an environmental chamber for the duration of the experiments. Embryos were maintained in 300 ml of seawater and gently aerated with approximately 60 bubbles/minute

for the entire incubation period. The combination of a high humidity and low rate of aeration resulted in minimal evaporative loss. Dissolved O_2 remained constant at 10-11 mg/L/L.

Transport Effects

In order to control for possible transport effects, ripe Puget Sound herring (PS) were transported to the University of Washington and spawned in the environmental chamber as described above. These were then incubated in parallel with the Prince William Sound embryos, exposed similarly to WSF and compared for differences in survival time and developmental abnormalities. The experiment was designed to detect any effect of prolonged transport.

Preparation of WSF

To test the toxicity of water soluble components of oil, a water soluble fraction was prepared by shaking 40 ml of oil with one liter of 29 ppt synthetic seawater in a 2L separator funnel at 8°C for 15 minutes, then allowing the mixture to separate in the funnel for 18 hours at 8°C. The funnel was tapped lightly several times to release oil droplets adhering to the glass, then the water layer was drained off into a glass-stoppered bottle and used as a stock for exposure dilutions. The seawater removed from the separator funnel was designated 100% WSF, and used as a stock from which dilutions down to 0.1% were made. A new stock was prepared every 48 hours

for the duration of the embryo exposures (approx. 21 days).

Chemical analyses of WSF were based on the total peaks observed for low molecular weight (C6-C12) gasoline range hydrocarbons, and high molecular weight (C12-C28) diesel range hydrocarbons. Low molecular weight (LMW) samples were analyzed by GC/FID (Purge & Trap) and the high molecular weight (HMW) samples by GC/FID, both modified U.S. Environmental Protection Agency method 8015. Because LMW values decreased with time, the HMW samples were used to convert "% WSF to real values corresponding to mg/L (ppm) of dissolved petroleum hydrocarbons. Analysis of three replicate extractions demonstrated that 100% WSF contained 9.67 mg/L (ppm) of HMW hydrocarbons.

Developmental Stage Sensitivity

To establish specific developmental stage sensitivity to WSF, 24 hours after fertilization, and every 24 hours thereafter, a different group of embryos was exposed to WSF for 36 hours. This resulted in four groups of embryos being exposed to WSF at 24, 48, 72 and 96 hours post fertilization. Three concentrations of WSF were used (25%, 50 & 100% WSF), which corresponded to 2.4, 4.8 and 9.7 ppm Prudhoe Bay crude oil. After the exposure period was complete, the embryos were washed free of adhering oil and incubated in flowing natural seawater until they hatched. The exchange rate was 6-8 times per hour, ensuring adequate aeration and removal of excreted metabolites.

Upon arrival at the University of Washington the Prince William Sound embryos were placed into 300 ml of WSF ranging from 100% to 0.1% (9.67-0.009

ppm). The WSF was changed every 48 hours for 18 days. Larvae were then allowed to hatch into clean seawater so that only embryotoxic effects would be measured.

Genotoxic Damage

Chromosome and mitotic damage was evaluated by examining mitotically active tissue from newly hatched larvae which had been exposed during the EC₅₀ determinations. The number of mitoses and the number of abnormal anaphase-telophase cells in each of the treatment groups were evaluated from germinal layer between the muscle cells and developing ray structure of the larval pectoral fin.

In Situ Embryo Exposures

Field deployment occurred in late April of 1991 in Prince William Sound. Slides containing newly fertilized eggs were placed into PVC cassettes and placed at two depths each of two oiled and two unoiled sites. The cassettes were retrieved from the field 10-12 days later and returned to the University of Washington. Embryos were in the environmental chamber in clean aerated seawater within 8 hours after being retrieved. Each exposure site received 250-350 eggs. Exposures occurred at 10 sites within Prince William Sound designated "C" and "O", with deployments below the mean low water mark at - 5 ft and - 15 ft.

(In Vitro) Embryonic Stage Sensitivity

The earliest exposure periods and highest WSF concentrations had the greatest effect on hatching success of herring embryos. The 24 and 48 hour developmental stages were the most severely affected with a 20-45% embryo mortality relative to the controls. By 96 hours post fertilization, embryo survival

increased and was 80-100% of the control levels.

Increasing concentrations of WSF produced an increase in the percent of abnormal larvae, ranging from 10% at 2.5 ppm to 65% in 10 ppm WSF. The exposed embryonic stage however, had no detectable influence on normal larval development. Abnormalities evaluated were scoliosis, lordosis, cranial malformations and optic deformities.

EC₅₀ Determination for WSF

Observations on the mean-hatching-day post fertilization revealed that embryo exposure to concentrations >0.242 mg/L WSF resulted in embryos hatching 4 to 5 days earlier (mean = 15 days) than did the controls and lower WSF concentration exposures (mean = 19 to 20 days). This concentration is one half the abnormality EC₅₀.

Continuous embryo exposure to WSF of crude oil had little or no effect on embryo survival or live hatch, but did significantly increase the percent of physically defective larvae. Physical deformities included spinal deformity (scoliosis or lordosis), optic malformations, mandibular malformations and an enlarged pericardial region. These defects appeared not to be pathognomonic so were all considered together for the purpose of this study (e.g. total abnormal larvae).

The abnormality data were analyzed using EPA's Probit Analysis Program for data from Acute and Short-Term toxicity tests with aquatic organisms (EPA Biological Methods Branch, Cincinnati, O.). An EC₁ to EC₉₉ curve was generated which shows an EC₅₀ of 0.432 mg/L of the high molecular weight components of WSF. The EC₁ was 0.078 mg/L and the EC₉₉ 2.39 mg/L. By comparing these values to those observed in the field, it

should be possible to predict the number of larvae which would be affected following exposure to a specific concentration of WSF *in situ*.

Normal untreated larvae were 71% heavier (2.4 mg/20 larvae) than the untreated abnormal larvae (1.4 mg/20 larvae). Normal larval weights decreased as the WSF concentration increased, but abnormal larval weights remained constant as the WSF concentration increased. Because many of the larvae were so severely deformed, it was not possible to obtain accurate lengths. Consequently, this measurement was abandoned in favor of the more consistent dry weight measurement.

Genotoxic Damage

Examination of mitotically active cells revealed that mitotic activity was significantly reduced at 0.24 ppm WSF and greater. Chromosome abnormalities micronuclei were significantly increased at and above WSF concentrations of 0.01 ppm, well below that which produced grossly visible physical abnormalities.

The mean number of embryos hatching in the control (C) group was significantly lower than in the oiled (O) group ($p < 0.01$; t Test). There was also a significantly greater number of normal larvae hatching from the "C" group (63.3%) than from the "O" group (51.3%) ($p < 0.01$; t Test). This response is similar to what was seen in the *in vitro* embryo exposures. It will be necessary to obtain analytical data on the chemical contaminants on site at the time of exposure before any correlation can be made between *in vitro* and *in situ* responses.

A dry weight analysis of larvae hatching from the two exposure groups showed that the mean weight of both normal and abnormal larvae from the

"C" group were significantly higher than the weights of the "O" group ($p < 0.01$; t Test).

The mean % hatch was 73 ± 11 for Prince William Sound and 65 ± 9 for Puget Sound embryos, and identical values for gross abnormalities, indicating no significant effect resulting from transport of the fertilized eggs from Alaska to Washington.

Conclusions

1. Early developmental stages (24-48 hrs post fertilization) appear to be the most sensitive to the effects of Prudhoe Bay WSF. The concentration of WSF and not developmental stage however, influenced the production of grossly abnormal larvae. Consequently, under natural exposure conditions it might be expected that early exposure of embryos would result in higher mortality while the concentration to which they were exposed would result in increased numbers of abnormal (non-viable larvae).
2. EC_{50} : The experimental EC_{50} of Prudhoe Bay crude oil WSF for Pacific herring embryos exposed for their entire incubation period was 0.43 ppm. By comparing this value with the concentration of WSF found following the Exxon Valdez oil spill, it should be possible to determine the total loss of herring embryos from oiled sites within Prince William Sound.
3. Genetic Damage: Significant damage to chromosomes and mitotically active cells in newly hatched herring larvae occurred at concentrations well below the EC_{50} , indicating that sublethal long-term damage had occurred and may not become apparent for several generations.
4. Hatching dynamics: Precocious hatching occurred at concentrations of WSF at or above 5% (0.484 ppm), with larvae hatching 4 to 5 days early. Alteration of mean hatching day frequently occurs in fish exposed to a wide range of pollutants. Early hatching may produce weak ill-prepared larvae which are more vulnerable to predation than those hatching after a normal incubation period.
5. Physical defects: The most obvious effect of WSF on developing herring embryos was the induction of physical defects in live-hatched larvae. Specific defects were not pathognomonic so were not reported separately. The LC_{50} for total abnormalities was 0.432 ppm with no normal larvae being present at or above 0.96 ppm.
6. Larval weights: A reduction in dry weight appeared to be related to exposure to WSF only in normal larvae. There was no correlation between abnormal larval weights and exposure to WSF in vitro. Not enough data is available to properly interpret the differences in larval weights observed at the oiled and unoled sites in Prince William Sound.



Egg-larval Mortality of Pacific Herring in Prince William Sound, Alaska, After the Exxon Valdez Oil Spill

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The Exxon Valdez oil spill of March 24, 1989, was followed 2.5 weeks later by spawning of the local stock of Pacific herring (*Clupea pallasii*). The effect of the spill on herring may have been restricted to eggs because growth and mortality of herring larvae captured 1 to 5 km offshore of the herring egg beds in 1989 were not significantly different between oiled and non-oiled areas (McGurk et al. in press). This study tested the hypothesis of an egg effect by comparing *in situ* egg-larval mortality between two oil-exposed areas and two non-exposed areas.

Two of the four major herring spawning sites in the Sound, the North area centered on Fairmount Island and the Northeast area centered on Tatitlek Narrows, were classified as non-oiled because the oil slick never contacted them. The other two spawning areas, the Naked Island archipelago and the northern tip of Montague Island, were classified as oiled.

Egg-larval mortality (Z , day⁻¹) was the ln-transformed ratio of larval density at hatch (NL , number m⁻² sea surface) to mean egg density (NE , number m⁻² spawning bed), divided by the duration of the egg-larval period (t , days), i.e. $Z = -(1/t)\ln(NL/NE)$.

Density of larval herring at each of the four areas was measured by weekly oblique plankton tows to 30 m depth from May 7 to June 22, 1989. Herring larvae were sorted from the zooplankton, counted and density was corrected

for net avoidance by larvae. A linear regression of $\ln(NL)$ on age was fit to the descending right-hand limb of the catch curve for the major cohort at each plankton station. Larval density at hatch was the intercept of the regression at $t = 0$ days.

Egg density at each area was measured by SCUBA divers as part of ADF&G's annual herring spawn survey. Mean egg density was the geometric mean of all transects that could have reasonably been expected to contribute larvae to the pool sampled by the plankton nets.

Aerial surveys by ADF&G found that mid-points of the spawning period in the four areas ranged from April 11 to 13. SCUBA surveys of egg density were conducted about 10-16 days later. Hatch dates of the four major cohorts of larvae occurred 7 to 12 days after the mid-dates of the SCUBA surveys.

Z was significantly different between areas; the greatest Z , 0.598 day⁻¹, was measured at an oiled site on Montague Island and the lowest Z , 0.065 day⁻¹, was measured at a control areas in the North area. Mean Z in the two oiled areas, 0.410 day⁻¹, was three times higher than mean Z in the two non-oiled areas, 0.123 day⁻¹.

Unfortunately, spatial distribution of herring eggs differed between the four areas in such a way as to confound the interpretation of egg-larval mortality. For example, mean width of herring spawn was significantly greater at Montague

Island than at the North area, which implied that the Montague Island site had a lower beach gradient than that of the North area. Since egg loss due to scouring by waves decreases with increasing depth, low gradient beaches will tend to have greater scouring than high gradient beaches, all other factors being equal. Also, the distribution of herring spawn ranged from 48.8% subtidal in the Northeast area to 75.4% subtidal at Bass Harbor. Since intertidal eggs are more vulnerable to desiccation and bird predation than subtidal eggs, egg-larval mortality decreases with increasing percent subtidal eggs.

We conclude that this study offers tentative support to the hypothesis that the *Exxon Valdez* oil spill increased egg-larval mortality in oiled areas of Prince William Sound. It is not conclusive evidence for an oil effect because an un-

known portion of the between-area differences in egg-larval mortality may have been caused by between-area differences in natural egg mortality. We cannot eliminate this possibility because we have no independent information on natural egg loss rates in 1989.

To the best of our knowledge, this study is the first to estimate herring egg-larval mortality by combining measurements of larval density with direct measurements of egg density. It is valuable because it provides estimates of egg mortality that are independent of the mode of mortality.

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Larval Fish Distribution and Abundance in Prince William Sound and Resurrection Bay During 1989

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There is no documentation of larval fish species and distribution in Prince William Sound prior to the Exxon Valdez oil spill. In response to the 1989 oil spill, six 1-week cruises in Prince William Sound (April-November) and four 4-day cruises in Resurrection Bay (May-July) were conducted. Both oiled and control unoiled sites were sampled. Plankton tows were taken with a 1 m² NIO tucker trawl or a 1 m² MOCNESS with 505 μ mesh nets. Discrete depth increments were sampled to 100 m in Prince William Sound and to 250 m in Resurrection Bay.

Over 40,000 larvae were collected in Prince William Sound and in Resurrection Bay. Over the 8-month sampling season, the greatest proportion of fish larvae were captured during May in both Prince William Sound and Resurrection Bay. Of the larvae captured in May, walleye pollock (*Theragra chalcogramma*) was the major species, comprising 80% of the fish captured. Pollock larvae were well distributed around the Sound.

In May concentrations of pollock larvae were collected in both oiled (Montague Island, Knight Island, Main Bay) and unoiled sites (Orca Bay, Port Valdez). Most of the pollock larvae were captured in the upper 50 m of the water column. Size distribution of pollock larvae captured within Prince William Sound in 1989 ranged from 2.8 mm to 11.3 mm and ranged in age from 1 to 40 days. A bell-shaped distribution of lengths probably indicates only one

cohort. Larvae on the western side of the Sound were slightly larger, hence older, than those found in the middle and eastern portions of the Sound, perhaps indicating movement through the region.

Pacific herring (*Clupea harengus pallasii*) were captured in May, June and July, but not April, September or October. In May, 321 herring larvae were captured. Most (197) were at one oiled station south of Naked Island. The others were taken in oiled sites (Knight Island) and unoiled sites (east side of Hinchinbrook Entrance). In June, 1349 herring larvae were collected. Of those, 1175 were in oiled areas around Montague and Knight Island, with 765 larvae taken at one station near Green Island.

Very few larvae were collected in unoiled areas near Ester Island and eastern Prince William Sound near Hinchinbrook Island. In July there were few (56) herring larvae evenly distributed in oiled and unoiled areas. Like pollock, most herring larvae were in the upper 50 m of the water column. The herring larvae ranged in size from 6.0 to 23.2 mm.

We will relate the distribution of larval fish to the distribution of the oil spilled. We will use length/frequency analysis to follow cohorts over time and space in relation to the distribution of oil. A modified Graded Severity Index (GSI) will be used to measure morphological deformities in herring larvae due to oil

exposure.

This study demonstrates that at least minimal baseline data are needed for species' presence/absence seasonally so

that in the event of an oil spill or other catastrophic event we will be able to assess the effect on larval fish, and the ultimate effect on the fisheries.



The Prince William Sound Herring Recruitment Failure of 1989: Oil Spill or Natural Causes?

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In 1989, herring spawned in Prince William Sound 2-4 weeks after the T/V *Exxon Valdez* ran aground. The resultant oil spill contaminated some herring spawning beaches and may have affected food chains used by larval herring. The Alaska Department of Fish and Game (ADF&G) has routinely collected a variety of assessment information on the Prince William Sound herring stock since the early 1970's. We sought to use this information to determine whether the abundance of the 1989 year class of herring was different than might have been expected from historical data.

Information about the abundance of a year class of herring is first obtained when herring begin to return to spawn in their third year of life; therefore, 1992 was the first opportunity to assess the effect of the oil spill on the abundance of the 1989 year class. Stock assessment data routinely collected by ADF&G include age compositions of the catch and spawning populations, aerial survey estimates of biomass, miles of milt observed from aerial surveys, and spawn deposition survey estimates of biomass.

We used an age-structured assessment (ASA) model to synthesize all of the available stock assessment information into a single time series of historical abundance. In this paper the ASA model of Funk and Sandone (1990) was updated to include additional sources of auxiliary information, additional gear types, and natural mortality estimation;

we also extended the time series of data to include information through the spring of 1992. Our goal was to produce a historical abundance time series that "smooths" or averages the often-conflicting stock assessment information. We then sought to examine how the strength of the 1989 year class compared with that predicted from the spawner-recruit model developed from the historical abundances and environmental conditions during early life history.

In a similar herring population in Sitka Sound, sea surface temperature anomalies explained at least 40% of the variability in recruitment patterns (Zebdi 1991). Because herring year class strength in Prince William Sound is correlated with the year class strength of other herring stocks around the Gulf of Alaska coast, we also sought to compare the relative strength of the 1989 year class in Prince William Sound with that in Sitka Sound.

Our approach uses an ASA model which incorporates auxiliary information, similar to that used by Deriso et al. (1985). The ASA model estimates initial cohort abundances which best fit observed age composition and abundance information, after accounting for removals at each age and year. Deviations of model estimates from observations are ascribed to measurement error in the observations.

While our primary goal was to use the model to estimate the historical

abundance time series, the model also estimates natural mortality, maturity, and gear selectivities for purse seine, gillnet, pound, and food and bait fisheries, and a coefficient which relates miles of milt observed in aerial surveys to spawning biomass.

The ASA model begins tracking herring cohorts at age 3, the first year that a measurable proportion usually return to spawn. The survival model accounts for natural mortality and harvest processes with a difference equation which describes the number of fish in a cohort at each age and year. The survival model removes the catch at each age resulting from the spring purse seine and gillnet and pound fisheries, and the fall food and bait fishery. The number of fish in a cohort includes both mature and immature herring measured at a time after annulus formation but before the spawning run or spring roe fisheries. The biomass of herring spawning at each age and year was estimated in the ASA model from the survival model's estimated number of fish at age, weight at age sampling, and the proportion mature at each age. The model estimated the proportions mature at each age and the proportion mature at each was assumed not to change from 1973 to 1992.

The harvest of herring by age for purse seines, sacroe, gillnets, sacroe, pound, and food and bait fisheries was tabulated for the 1973 to 1992 period from ADF&G catch records. Observed numbers of fish in the catch for each gear were also converted to age composition (percent by age) for each gear, for comparison to age compositions estimated from ASA model. Gear selectivity was defined to include both the effects of immature fish not being present on the fishing grounds

(partial recruitment or maturity), and active selection or avoidance of certain fish sizes by the gear or fishermen's behavior. A logistic selectivity function was used for gears which were thought to have asymptotic selectivities (purse seine, pound, and food and bait); a gamma-type function was used for gillnet gear where selectivity might decrease at the older ages.

The volume of milt deposited by male herring each year was assumed to be proportional to the mature biomass. Since 1972, ADF&G aerial herring surveys have routinely recorded the miles of shoreline adjacent to milt discolorations visible in the water. A goodness of fit measure for miles of milt was developed by assuming that this linear measurement was directly proportional to the mature biomass. Spawn deposition surveys were conducted in 1984, and 1988-92. These surveys estimate biomass by back-calculation from the numbers of eggs deposited, using additional sampling to estimate fecundity and sex ratio. A goodness of fit measure for the ASA model was developed from the differences between ASA estimates of mature numbers at age and the spawn deposition survey estimates of numbers at age.

In addition to the time series of the catch by age, a relatively long time series of age compositions of the spawning population are available. Since 1984, age samples have been collected from spawning herring. Sampling effort was lower for years prior to 1984, and spawning age compositions were reconstructed primarily from purse seine catch samples from each area. Sample sizes were judged to be too small in 1973, 1974-78, and 1980-81 to reliably construct estimates of spawning age composition. A goodness of fit

measure was developed from these age compositions as the difference between the ASA model's estimated age compositions and those observed during ADF&G sampling.

A total sum of squares was computed by adding each of the component goodness of fit measures where each component was assigned an ad hoc weight. The ad hoc weights reflected our attempt to weight data equitably, but also incorporate some prior knowledge of our confidence in each component. The model estimates a total of 37 parameters: 23 initial cohort sizes, 10 gear selectivity function parameters, 2 maturity function parameters, 1 aerial milt survey biomass coefficient, and 1 survival rate parameter. The combined weighted sum of squares was minimized using nonlinear least squares techniques to estimate values for the 37 parameters.

Biomass estimates from the ASA model were relatively low (20,000–40,000 metric tons) during the 1970's and increased to higher levels (50,000–110,000 metric tons) in the 1980's. Recent trends in abundance, indicated by the age composition and aerial milt survey data, are different than the abundance trend from the spawn deposition survey. While the strong 1984 year class began dominating the age compositions in 1987, the spawn deposition survey biomass did not increase until 1990. The cause for this discrepancy is unknown, but it may indicate that spawn survey measurement error is much greater than anticipated.

While the design goal for spawn deposition surveys was a precision such that 95% confidence intervals would be $\pm 25\%$ of the true biomass value, the absolute deviations of spawn deposition survey biomass estimates from the ASA biom-

ass estimates averaged 48%. The ASA model reflected these inconsistencies by essentially scaling the biomass to a "smooth" of recent spawn deposition survey biomass estimates, while manipulating year class strengths to track trends in age composition data more closely. The estimated survival rate of 65% (equivalent to an instantaneous natural mortality rate of 0.43) was very similar to the midpoint of the range used by Funk and Sandone (1990). The ASA model estimated that, on average, one mile of milt from aerial surveys corresponded to be 821.2 metric tons of spawning herring.

Year class strength in Prince William Sound is characterized by occasional years of very strong recruitment. Beginning with the 1976 year class, these strong year classes have occurred every four years. The 1989 year class is among the smallest observed since the beginning of the data series in the early 1970's and resulted from one of the largest egg depositions. First quarter sea surface temperatures in 1989 were also relatively low and low sea surface temperatures tend to be associated with weak year classes.

However, corresponding data from Sitka Sound indicate that the 1989 year class there was not as weak. Because the 1989 year class has appeared in assessment samples only for a single year, the precision of the estimate of its abundance is not high.

Precision is further reduced because only a portion (approximately 25%) of a year class is recruited at age 3, and the proportion recruited varies somewhat from year to year. The precision of the abundance estimate of the 1989 year class will continue to improve with additional years of sampling. However, most of the

improvement in the precision of the abundance estimate for the 1989 year class will be realized by 1995.

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Adult Herring Reproductive Impairment Following the *Exxon Valdez* Oil Spill

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This project was designed to evaluate the reproductive potential of individual female herring which had been present as 1-year-olds in Prince William Sound at the time of the *Exxon Valdez* oil spill. Two groups of 25 females each were collected in Rocky Bay (Montague Island), a known previously oiled site and one group of 25 was collected from Boulder Bay, an unoiled site.

The target fish were 4-year-olds which had presumably been exposed to Prudhoe Bay crude oil 3 years earlier, at the time of the *Exxon Valdez* spill in 1989, and were returning as first-time spawners in 1992. Previous studies had shown differences in the development of embryos and larvae from oiled and unoiled sites, and it was not clear as to whether these were due to site specific differences or previous exposure of spawners (Kocan, 1991).

Running ripe females were collected by gillnet from one unoiled site in Boulder Bay on 4/11/92, and two previously oiled sites in Rocky Bay on Montague Island on 4/21 and 4/22. Because of spawner distribution within Prince William Sound in 1992, bad weather and transportation restraints, we were not able to collect two groups each from oiled and unoiled sites as originally planned.

Males and females were sorted and packed in a cooler to maintain their ambient temperature and returned to Cordova by air where they were given

ID numbers and artificially spawned onto glass slides. Approximately 200-400 eggs were obtained from each female for the purpose of this study. These were fertilized with pooled sperm from five males.

Following spawning, length and weight were determined, scales collected for age determination, and tissues collected and preserved for histopathology. The entire process from field collection of fish to completion of spawning was completed within six hours and no differences in fertilization or embryo viability was noted during this period. The eggs-on-slides were then placed into a transport chamber, gassed with pure O₂ and transported by air to the University of Washington Friday Harbor Laboratories for incubation. When embryos arrived at Friday Harbor they were 24 hours old and in early blastodisc stage of development. Water temperature was maintained at 5-6°C from time of collection until their arrival at Friday Harbor, where it was allowed to rise to local ambient water temperature of 9°C over a 6 hour period.

Developing eggs from each female were placed in individual flowing seawater chambers for incubation and were examined for fertility, embryo survival (hatching) and, following hatching, for physical defects which would affect their viability. Defects were categorized as follows: (1) spinal deformities; (2) craniofacial deformities; (3) optic abnormalities; and (4) pericardial edema. These

categories were pooled and evaluated in relation to female:female variability and oiled vs. unoled sites.

It was anticipated that this study would clarify whether previously observed differences between oiled and unoled sites were the result of site differences or oiling differences, as well as give some insight into the natural hatching variability and larval viability which might be expected from individual females of a particular age class.

There was no significant difference in fertility rate among the three groups (3-6%). There was however, a significant difference in hatching and abnormal larvae between the oiled and unoled sites. The percent total hatch observed in Boulder Bay, the unoled site, was $56\% \pm 13$, while the two spawning groups collected in Rocky Bay, the oiled site, were significantly lower, at $19\% \pm 9$ and $39\% \pm 10$ respectively ($P < 0.01$; t Test). These values are based on the total number of live larvae produced relative to the number of eggs fertilized, and fall within the range of previously reported studies (Rosenthal & Hourston, 1982 and Hansen et al., 1985).

The percent of normal larvae produced from these same eggs was $35\% \pm 14$ for Boulder Bay and $14\% \pm 10$ and $27\% \pm 6$ respectively for Rocky Bay ($P < 0.01$; t Test). The distribution of abnormalities appeared to be random, with no single abnormality category being pathognomonic for oil exposure.

Mean female herring lengths for the three groups were 199, 202 and 206 mm respectively, and regression analysis demonstrated a linear relationship for length and weight for females from all three collection sites ($R^2 = 0.85$). There was no correlation between female length

and abnormal larvae in any of the three groups.

The females collected from Boulder Bay, a site not oiled during the Exxon Valdez spill produced significantly more live larvae and more normal larvae than did two groups of female herring collected at Rocky Bay, a lightly oiled site. There was no apparent difference in the weight, length or fertility rate among females from any of the three sites.

These results are in agreement with previous data collected in Prince William Sound which showed oiled sites to be less suitable for normal herring embryo development than were unoled sites. There is however, still the question of whether the observed effects are the result of (1) naturally occurring site specific differences, (2) residual oil at each site, or (3) previous exposure of spawners to oil or oil products during or subsequent to the 1989 Exxon Valdez oil spill. Each life stage of the herring studied in situ and in vitro since the spill, has demonstrated some toxic or pathologic response. Whether these are related to the observed poor performance of developing herring embryos is not known, but it is possible that each of them separately or in combination could have contributed to the reduction in embryo survival and subsequent viability.

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Summary of Known Effects of the *Exxon Valdez* Oil Spill on Herring in Prince William Sound, and Recommendations for Future Inquiries

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The herring population in Prince William Sound, because of its size and biomass, is a critical food source for many avian, mammalian, and subtidal predators, is an important subsistence food, and a target for a multi-million dollar commercial fishery. Unfortunately, the grounding of the T/V *Exxon Valdez* on March 24, 1989 and its resulting 11-million-gallon oil spill coincided with the annual spring migration of herring spawners to nearshore staging areas. Over 40% of areas used by herring to stage, spawn, or deposit eggs and over 90% of areas needed for summer rearing and feeding were exposed to crude oil.

From 1989 to 1992, the Alaska Department of Fish and Game, the University of Alaska, and the National Marine Fisheries Service conducted studies describing the damage done to Prince William Sound herring as a result of the spill. Herring embryos (live eggs) and hatched larvae were studied in the field and in the laboratory, juvenile herring were trawled during the summer of 1989, and adults were enumerated and collected for various laboratory tests. The total numbers of herring spawning were estimated annually. Since the herring population is composed of up to ten year classes, or fish born in different years, the age composition was determined annually through an intensive sampling program. In the laboratory, damage observed in eggs, larvae, and adult herring was related to known concentrations of oil in

dose-response experiments. In 1992, eggs were collected from adult female herring that were artificially spawned and reared separately for each female. The hatching success of these eggs and larval abnormalities were measured to look for residual effects of oil on the adult herring population.

Potential injury to the herring population in Prince William Sound was studied in three life history stages. First, the early life stages, from egg to hatched, free-swimming larvae were studied for each year from 1989 to 1991. Second, larval-juvenile fish were collected by trawl and studied in 1989. Third, adult herring were collected from 1989 to 1992 and analyzed.

Damage summaries for each of the three categories were synthesized separately because important biological information needed to link the information between life stages was missing. Information needed to link life stages and create a complete population dynamics model include: (1) the number of their stocks and their distribution in Prince William Sound; (2) rates of herring immigration and emigration in Prince William Sound; (3) how environmental factors influence survival of larvae, juveniles and thus affect recruitment; and (4) how population levels affect survival of young fish (density-dependent factors).

Damage to Prince William Sound herring is described in detail in the indi-

vidual project reports. Although egg mortality was slightly elevated in oiled areas, lethal and sublethal genetic damage and physical abnormalities were much greater in oiled areas than in non-oiled areas in 1989. Injuries were more common and more severe in oiled than un-oiled areas, and in all aspects, injuries declined in 1990 over 1989. Most of the damages documented in 1989 and 1990 were similar to impacts recorded from other oil spills and laboratory experiments. Genetic damage (anaphase aberrations) was the most sensitive measure of oil damage, measurable even at the lowest concentrations of oil used in the laboratory studies. Survival from egg to free-swimming larvae was three times greater in un-oiled areas, however, environmental and biological factors confounded the results. A model to estimate total damage from egg to free-swimming larvae and combining all the effects measured in 1989 to 1991 will be completed soon.

The juvenile fish trawl survey confirmed that herring larvae hatching in 1989 followed the same path as the oil trajectory through Prince William Sound which may have further impacted that year class. Measurements of physical abnormalities, chromosomal breakages, and tissue or histopathological injury are currently being summarized to estimate the extent of the damage.

Internal tissue damage found in adult herring resulting from direct exposure to oil suggests that this toxic event may have weakened the fish's ability to resist diseases and parasites. In addition, certain gut-dwelling parasites were found to have migrated deeper into the adult herring musculature, possibly a response to oil in the digestive tract (Moles et al., in press). Juvenile pink salmon were

stunted by ingestion of oil (Moles et al., in press), so a similar effect may have occurred with juvenile herring.

The absence of three-year-old herring, born in 1989, sampled from the spawning population in 1992 and possible reproductive impairment of the four-year-old herring, born in 1988 may indicate further effects from the spill. Environmental factors affecting the 1989 year class confound a clear understanding of the oil spill impact on the adult population.

The 1988 year class was one and a half years old during the spill year. The hatching success of eggs collected from four-year-olds at an oiled area was less than half that of eggs collected in an un-oiled area, even though no oil remained at the sites. The 1988 year class currently dominates the Prince William Sound herring spawning population and if a portion of them cannot reproduce successfully, this may affect the population in the future.

The reproductive study conducted in 1992 was a pilot project initiated to measure if differences in reproductive rates could be detected between areas. Further study might resolve whether the reproductive differences measured were due to oil or were due to natural biological variation.

The 1988 year class as well as others (e.g. 1984 and 1989 year classes, if available) should be tracked in future studies, to look for differences between ages. Until some of the linkage data between life stages and environmental parameters affecting the Prince William Sound herring population are understood, it will be difficult to project exactly how the potential reproductive impairment measured in adults spawning in 1992 will affect the future population.

With a better understanding of fac-

tors affecting survival and recruitment, a population simulation model could be constructed and used to predict damage in the case of another spill or toxic event. Some of the basic information needed to study the natural recruiting process is present and is ready for a modeling exercise. This model could also incorporate the reproductive impairment information and be used to predict future impacts.

Natural restoration of the Prince William Sound herring population is probably the best tool available to mitigate oil damage to herring stocks. Because artificial propagation of juvenile herring is not currently being done and because it is difficult to enhance spawning substrates due to movements of the exact spawning locations of herring from year to year, it is probably not practical to use these hands-on techniques.

Since herring are harvested in large numbers by humans, alteration of human use could be an effective restoration tool. However, using alteration of human use or fisheries management to avoid oil-damaged stocks or to mitigate damage can be successful only if precise stock assessment tools or a good understanding of the Prince William Sound population exists. At this time, we do not possess a precise enough description of the Prince William Sound stock or an estimate of the total damage accrued to accurately adjust fishing pressure and protect future populations from a possible oil-induced decline. In addition, as an agency mandated by state law and policy, we cannot make adjustments to the current management plan without sufficient justification because we are held liable for resulting losses.

From my observations while implementing the herring program over the

last four years, I have some recommendations concerning future planning in spill response, damage assessment and restoration that could improve the process.

First, a response plan is sorely needed that incorporates all the agencies (with cooperative agreements in place), puts appropriate expertise in advisory roles available to all levels of responders and planners, that defines appropriate chain of command and roles for each staff person involved, and that sets some emergency administrative procedures in place.

Secondly, a similar plan for damage assessment is needed to coordinate responders and researchers, make experts available to assist in survey design, and include baseline information (from previous research and ongoing monitoring) from which to build studies. As part of summarizing damage assessment, a panel of experts would conduct an environmental modeling exercise to link damages between species, guide research, and estimate damage for the legal process.

Finally, restoration planning should start the same day as the spill and coordinate intimately with damage assessment using the results of the environmental monitoring exercise and drawing from technical information available on effective restoration techniques. Out of the restoration planning process, a list should be constructed of needed studies or enhancement projects so that proposals could be solicited from interested researchers and resource managers. Norcross (1993) refers to Norwegian response plans that accomplish many of the goals listed above.

In lieu of having none of the above available at the time of the *Exxon Valdez* spill and a very primitive understanding

of the Prince William Sound ecosystem and detailed population damage information, I recommend, as an absolute minimum, that we proceed rapidly with habitat protection and monitoring before all of the settlement monies are spent. Habitat protection will prevent the exacerbation of oil spill damage (known and unknown) and a comprehensive monitoring plan will provide baseline information needed for the planning process

described above. Much time and money could be saved in the future if some were invested now in planning.

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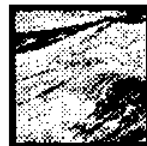
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Rehabilitating Oiled Sea Otters and Other Fur-bearing Marine Mammals: Lessons from the Exxon Valdez Oil Spill

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The rescue and rehabilitation of 225 oiled sea otters during the *Exxon Valdez* oil spill remains the largest effort of its kind ever attempted (Williams and Davis, 1990; Davis and Styers, 1991; Davis and Williams, 1991a; Davis and Williams, 1991b; Williams et al., 1991;). Although the number of sea otters saved had little effect on the overall sea otter population in Alaska, the experience contributed enormously to the wildlife rehabilitation community's understanding of what is needed to successfully rescue and treat sea otters after an oil spill. In addition, the information we gained may one day prove invaluable for species or populations which are threatened or endangered. This information is being incorporated into a new handbook on the emergency care and rehabilitation of oiled sea otters and other fur-bearing marine mammals. This practical guide will provide information on rehabilitation facilities, management and personnel, capture, transport, triage, cleaning, clinical care, pathology, toxicology, nutrition, and the husbandry of adult, juvenile and pregnant sea otters. With contributions from over twenty authors, this book will provide the most up-to-date information on sea otter oil spill contingency planning and response.

Sea otters are perhaps the most vulnerable of all marine mammals to the detrimental effects of oil spills. This results from their dependence on fur for thermal insulation in the cold marine

environment. Exposure to oil eliminates the air layer trapped within the fur and reduces the thermal insulation of the pelt by 70% (Williams et al., 1988). Without this insulation, the otter has a very limited ability to increase metabolism and the intake of food energy to counteract the increased heat loss (Davis et al., 1988). As a result, a heavily oiled sea otter must leave the water (where it cannot feed), or it will become hypothermic. Whether it remains in the water or hauls out on land, a severely oiled otter will eventually die.

Sea otters are also vulnerable to petroleum hydrocarbon toxicosis (poisoning) during an oil spill. This results primarily from their behavior of resting and eating on surface of the water where oil is concentrated during a spill. Sea otters may inhale volatile petroleum hydrocarbons and absorb heavier fractions through their skin. The ingestion of oil may result when otters lick their fur during grooming and when they bring their food to the surface to eat. Heavily oiled sea otters must be captured and treated quickly if they are to survive.

A comprehensive sea otter oil spill contingency plan must include the following components: (1) facility design and maintenance; (2) administration and staffing; (3) the location and capture of oiled otters; (4) air and ground transportation of the otters to the rehabilitation facility; (5) cleaning and rehabilitation; (6) animal husbandry; (7) veterinarian

care and pathology; (8) short and long duration holding; (9) release and monitoring of rehabilitated otters; (10) liaison with responsible government agencies, community groups and the press; (11) feeding and housing the staff.

The plan must make provision for seasonal differences in weather and sea conditions. A facility for treating 200 otters will require a staff of 100-200 professionals and volunteers. The number of capture boats required will depend on the size of the spill and the number of otters at risk. The boats are most efficiently used when they remain on site, and the captured otters are transported to the rehabilitation facility by helicopter. A core staff of experienced animal care specialists and veterinarians can be assisted by a less experienced paid staff and by volunteers provided that they are adequately trained to care for the otters. Throughout the process of capturing, cleaning and rehabilitating the oiled otters, good communication must be maintained with responsible federal and state government agencies.

A plan to release rehabilitated otters should consider the duration and severity of the effects of the oil on the marine habitat and whether long-term monitoring of the otters by radio-telemetry is necessary. Tracking radio-telemetered otters is expensive and the data can be difficult to interpret if the otters leave the survey area. In addition, the detrimental effects of surgical implantation of radio telemeters into otters that have just recovered from oiling may further bias the estimated survivorship of rehabilitated animals. Of the 45 rehabilitated sea otters that received abdominal radio implants after the *Exxon Valdez* oil spill, 14 died within the first year. The cause of death for these otters remains unknown.

Many others have not been located and their fate is uncertain.

In order to care for more than 20 oiled sea otters at one time, a rehabilitation facility must be designed to move animals efficiently through the various stages of rehabilitation. Depending on the level of exposure, an oiled otter may require a few days to several months to complete the rehabilitation process. Once the otters have regained their health, they should be moved immediately to a pre-release facility or returned to the wild. By designing the rehabilitation facility as a flow-through system, the number of otters that can be treated during an oil spill is much greater than the holding capacity of its pens and pools.

The most effective and cost efficient way of caring for large numbers of oiled sea otters is to concentrate resources and expertise in regional rehabilitation centers. Each regional center should be strategically located in an area where sea otters are abundant and at risk from an oil spill. By using a helicopter to transport oiled otters from the capture boats to the rehabilitation facility, each regional center can service an area within a 500 mile radius (i.e. within a five hour helicopter flight). Beyond the 500 mile radius, trained personnel and mobile facilities are needed to medically stabilize the newly captured otters before they are flown to the regional center.

Indoor space at a regional rehabilitation center is required for: (1) documenting and stabilizing newly arrived otters; (2) cleaning and drying otters; (3) critical care; (4) holding pens with pools; (5) food preparation and cold storage; (6) a veterinarian clinic and necropsy room; (7) a nursery for pups; (8) administration and security; (9) an area for the staff to dress, eat their meals and receive train-

ing; (10) dry storage. Outdoor space is required for holding pens with small pools, large pools, seawater filtration, and storage. A facility to treat 200 otters at one time requires about 13,500 ft² of indoor space and 42,000 ft² of outdoor space. Essential amenities include hot and cold fresh water, a sea water supply (2000 gal/min), electricity, and sewage. Large holding pools should provide each otter with 20 square feet for swimming and 8 square feet of dry deck for haul out. Because sea otters may not be released immediately after rehabilitation, a pre-release facility, consisting of seawater holding pens, is needed. Holding pens in the pre-release facility should be large enough for the otters to swim and dive (at least 20 ft. long, 10 ft. wide, 5 ft. deep) and have good seawater circulation.

Transporting sea otters over long distances is stressful. For otters that have been exposed to oil, this stress can cause death or seriously complicate medical conditions. Mobile triage units are beneficial in certain cases because they allow staff to stabilize the animals medically before they are flown long distances to the regional rehabilitation center. Mortality can be reduced significantly if mobile units are employed when large numbers of otters must be captured more than 500 miles (five hour helicopter flight) from the regional center. These mobile units, by necessity, are self-contained (including electrical generators and hot water) and transportable by truck, boat, large helicopters, or fixed-wing aircraft (C-130 cargo plane); the latter may be necessary for areas inaccessible by road such as the Aleutian Islands. A mobile facility should consist of a trailer (10 ft. wide, 30 ft. long) that is divided into three functional sections for triage and stabilization, food preparation and a

small veterinary clinic.

The key to saving the maximum number of oiled sea otters is through preparedness and rapid response. The most critical time for saving oiled otters is during the first two weeks of a spill. This is true for both preemptive capture before oiling and for otters that have become oiled. If contingency plans are inadequate and the first few weeks are spent assembling a response effort (as was the case with the *Exxon Valdez* oil spill), then the number of otters saved will be greatly reduced. At this time, our understanding of how to rehabilitate oiled otters will not be the factor which limits survivorship. The limiting factors will be the time involved to mobilize the capture effort, build appropriate rehabilitation facilities, and train rehabilitators.

To be most effective, a rescue effort must be able to respond within in six hours. This is only possible if the capture boats are pre-identified and under contract, the rehabilitation facilities are already built, and the staff has been trained before the spill occurs. Unfortunately, the State of Alaska and the oil industry are still reluctant to invest the amount of money needed to build a regional rehabilitation center that will enable a rapid response. Ironically, investing in a rehabilitation center and training programs before a spill occurs will actually reduce the cost of caring for oiled sea otters by 50% to 80%. Given that the public will demand a sincere and conscientious attempt to save oiled sea otters, preparedness and a rapid response will not only save more otters, it will save money too.

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The Efficacy of the T/V Exxon Valdez Oil Spill Sea Otter Rehabilitation Program and the Possibility of Disease Introduction into Recipient Sea Otter Populations.

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Prior to the massive oil spill caused by the wreck of the T/V Exxon Valdez, it had already been established that the sea otter (*Enhydra lutris*) was extremely vulnerable to oil contamination. Thus, after the spill, several hundred sea otters were captured and brought into centers that were established in order to wash them, and to provide them with medical and other supportive treatment.

Some of these sea otters were obviously contaminated with oil, whereas others were not. Many of the sea otters that survived such treatment were eventually released into wild (recipient) populations in eastern Prince William Sound and along the Kenai Peninsula.

The original goal of the study reported here was to obtain data on the survival of these sea otters in order to provide information about the efficacy of the rehabilitation strategy. Thus, forty-five sea otters (18 males and 27 females) were selected from otter centers during July and August 1989 and instrumented with surgically implanted radio-transmitters using protocols identical to those used on sea otters in previous and subsequent studies.

Of the sea otters at the treatment centers, an attempt was made to choose the healthiest individuals for inclusion in this study. While a few of the otters selected had been classified as heavily oiled ($n = 3$) at the time of admission to the centers, the vast majority were lightly oiled ($n = 28$), the amount of oil on several was termed medium ($n = 9$), and a few were classified as being unoiled ($n =$

3) (the oiling status of 2 otters in this study was not recorded by treatment center personnel, but they were not washed, and, thus, were probably unoiled). These forty-five radio-instrumented sea otters were released into eastern Prince William Sound during summer, 1989, and they were monitored regularly for over two years.

The term rehabilitate means to restore to customary activity or to a former state. The findings presented and discussed in this paper suggest that the combination of measures undertaken in an attempt to aid sea otters after the T/V Exxon Valdez oil spill did not result in the true rehabilitation of the surviving otters.

Our viewpoint is that captivity, in and of itself, poses serious dangers to the specific otters brought in, to the population exposed to capture procedures during an oil spill, and to the population into which the otters are released. Factors contributing to this risk are (1) stress; (2) disease, which risks the captive population and eventually through release, the wild population; (3) separation of mother-pup pairs; and (4) disruption of the natural learning processes of young animals.

In this paper we provide data on the survival rates of the animals from the treatment centers after their release back to the wild. Additionally, we provide data on survival rates of adult sea otters in the wild eastern Prince William Sound population into which they were released.



Clinicopathologic Alterations in Oiled Sea Otters Dying Acutely in Rehabilitation Centers

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Following the *Exxon Valdez* oil spill, a major effort was made to treat oiled sea otters in rehabilitation centers in an attempt to return them to the wild. There were 347 sea otters brought to rehabilitation centers; of these, 116 died, 94 (81%) within 10 days of arrival at the centers. Clinical records of 23 otters dying during their first 10 days within the centers have been reviewed in an attempt to define the clinical syndromes associated with these acute deaths. The 23 otters were selected on the basis of completeness of clinical records and availability of gross and histopathologic necropsy reports.

On the basis of appearance at the time of presentation at the centers, 7 of the 23 otters were classified as heavily oiled (greater than 60% of the body covered by oil), 7 of the 23 were classified as moderately oiled (30%-60% oil coverage), and 9 of the 23 were classified as lightly oiled (less than 30% oil coverage).

The most common terminal clinical syndrome seen regardless of degree of oiling was shock characterized by hypothermia, lethargy, and often hemorrhagic diarrhea. Shock was rarely observed at the time of presentation but in heavily and moderately oiled otters generally developed within 48 hours of initial presentation. In the lightly oiled otters, shock generally occurred during the second week of captivity.

A high proportion of otters in all three groups died with convulsions. Four heavily oiled, 4 moderately oiled and 3

lightly oiled otters were in seizure at or near the time of death. Anorexia was also a common problem, occurring in 6 of the 23 otters.

Complete blood counts and serum chemistry collected near the time of death were available for all 23 otters although not all analytes were measured in every otter. These data were compared to reference ranges established for normal otters from southeast Alaska. Alaskan reference ranges were very similar to those established for the California sea otter (Williams and Pulley, 1983; Williams et al., *In press*). There is little in the literature regarding the interpretation of abnormal clinical laboratory findings in sea otters. Abnormalities were therefore interpreted according to conventions used for the interpretation of laboratory data in dogs and other carnivores/omnivores (Duncan and Prasse, 1986) with consideration of both clinical history and necropsy findings.

The most common hematologic abnormalities from all groups included leukopenia characterized by neutropenia with increased numbers of immature neutrophils (degenerative left shift), and lymphopenia. Anemia was also relatively common. Degenerative left shifts reflect severe inflammation and are commonly observed in animals suffering from diarrhea with bowel stasis, a possible scenario in these otters. Lymphopenia reflected systemic stress associated with increased endogenous

glucocorticoid levels and resultant sequestration and possible destruction of circulating lymphocytes. Cause of the anemia could not be determined from the available data.

Principal clinical chemistry abnormalities included azotemia, hyperkalemia, hypoproteinemia/hypoalbuminemia, elevations of serum transaminases indicating hepatocellular leakage, and hypoglycemia.

Azotemia was equally prevalent in otters in all three groups. Urine specific gravities were not available to help differentiate prerenal from renal azotemia. However, necropsies did not reveal significant renal lesions, suggesting that shock and associated hemorrhagic diarrhea led to prerenal azotemia as a result of reduced renal perfusion. In the few animals that probably had true renal azotemia (serum urea nitrogen values of greater than 200 mg/dl), it is likely that long-standing reduced renal perfusion eventually led to primary terminal renal injury.

Hyperkalemia and hypoproteinemia/hypoalbuminemia, while less consistent than azotemia, were probably also related to diarrhea and shock. Hyperkalemia was probably at least partially the result of release of potassium from dying cells. Acidosis, a common accompaniment of diarrhea, also causes hyperkalemia. Hypoproteinemia and hypoalbuminemia were probably the result of protein loss in the diarrhea fluid.

Elevated serum transaminases indicating hepatocellular leakage were only slightly less common than azotemia. Hepatocellular leakage may have been a reflection of primary hepatotoxicity but was more likely a nonspecific change

associated with anorexia. In anorexia, there is mobilization of tissue stores of fat to liver which in turn results in increased cell membrane permeability with leakage of transaminases into the blood. In the otters, elevated transaminases correlated most frequently with hepatic lipidosis histologically. Hypoglycemia probably also resulted from anorexia and may have been the cause of many of the terminal convulsions.

Whether or not shock associated with hemorrhagic bowel syndrome was a direct primary effect of oiling or primarily an indirect effect secondary to confinement and handling in the rehabilitation centers is difficult to assess. Lightly oiled otters were as likely to die from shock as heavily oiled ones, suggesting that confinement was more important than direct exposure to oil. However, heavily oiled otters developed shock more rapidly and had greater numbers of laboratory abnormalities, suggesting that at the least, exposure to oil was an important predisposing factor.

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Histopathologic Lesions in Sea Otters Exposed to Crude Oil

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On March 24, 1989, the oil tanker *Exxon Valdez* ran aground on Bligh Reef in Prince William Sound, Alaska. The resulting spill of approximately 41.6 million liters of North Slope crude oil was the largest tanker spill in the history of the United States. In the months following the spill, over 1,000 sea otters from oil spill-affected areas are known to have died. The actual number that died was probably much greater.

The purpose of this study is to identify and describe histopathologic (tissue) lesions associated with crude oil exposure in sea otters and to discuss possible pathogenesis of the lesions. Materials available to us included tissues from oil-contaminated and uncontaminated otters that died in rehabilitation centers following the oil spill and tissues from otters that were found dead in the oil spill-affected area with external oil present. We also examined tissues from apparently normal sea otters from an area not contaminated by crude oil.

Following the oil spill, sea otters that appeared oil-contaminated, were in danger of becoming oil-contaminated, or were behaving abnormally were captured and taken to rehabilitation centers. Oil exposure was assessed by visual examination on arrival at the centers. Degree of oil contamination was graded according to the following criteria: oil covering greater than 60% of the body - heavily contaminated; oil covering 30-60% of the body - moderately contami-

nated; oil covering less than 30% of the body or light sheen on fur - lightly contaminated. If there was no oil visible, otters were considered uncontaminated.

Fifty-one oil-contaminated otters (14 males and 37 females) died in rehabilitation centers (Group No. 1). Six uncontaminated otters (three males and three females) died in rehabilitation centers (Group No. 2). Five otters (three males and two females) were found dead with external oil present (Group No. 3). Six apparently healthy sea otters (four males and two females) were killed by gunshot in an area not affected by an oil spill as part of unrelated research (Group No. 4).

In oil-contaminated sea otters that died in rehabilitation centers (Group No. 1), interstitial pulmonary emphysema was the most prevalent lesion, being present in 11/15 (73%) heavily contaminated, 5/11 (45%) moderately contaminated, and 3/20 (15%) lightly contaminated otters. Histologically, the lesion appeared as expanded areas of clear space with rounded contours within the interlobular septa. Occasionally, adjacent parenchyma was compressed or atelectatic.

Gastric erosions were seen in 2/14 (14%) heavily contaminated, 7/9 (78%) moderately contaminated, and 4/17 (24%) lightly contaminated Group No. 1 sea otters. Histologically, the erosions appeared as discrete areas of coagulative necrosis, measuring 1 mm to 3 mm in diameter, affecting superficial to mid-

level gastric mucosa. Variable amounts of hemorrhage and dark brown pigment produced by acid digestion of blood were present in the necrotic areas. Small numbers of neutrophils were sometimes scattered along the margins of the erosions.

Hepatic lipidosis was present in 8/16 (50%) heavily contaminated, 5/12 (42%) moderately contaminated, and 1/19 (5%) lightly contaminated Group No. 1 otters. The lesion was characterized by the presence of variably sized, usually multiple but occasionally single, round, sharply delineated, unstained intracytoplasmic vacuoles in periportal hepatocytes; in the more severely affected livers, midzonal and centrilobular hepatocytes also contained vacuoles. In oil red O-stained sections, the intracytoplasmic vacuoles stained red, indicating the presence of lipid.

The prevalence of renal lipidosis was somewhat less than that of hepatic lipidosis, being found in 10/42 (24%) Group No. 1 otters. All otters with renal lipidosis also had hepatic lipidosis. Microscopically, affected kidneys had single or multiple, variably sized, round, discrete, unstained intracytoplasmic vacuoles within proximal and distal tubular epithelium. The vacuoles stained red with oil red O.

Centrilobular hepatic necrosis occurred in 4/16 (25%) heavily contaminated, 3/12 (25%) moderately contaminated, and 4/19 (21%) lightly contaminated Group No. 1 otters. In affected livers, centrilobular hepatocytes had undergone coagulative necrosis characterized by pyknosis, karyorrhexis, karyolysis, and increased eosinophilia of cytoplasm with preservation of basic cell shape.

Of the 6 uncontaminated otters that died in rehabilitation centers (Group No.

2), 1 male (17%) had gastric erosions, 1 female (17%) had periportal hepatic lipidosis and multifocal hepatic necrosis, and 1 female (17%) had focally extensive hepatic necrosis.

Of the 5 sea otters found dead with external oil present (Group No. 3), 1 female had interstitial pulmonary emphysema, periportal hepatic lipidosis, and renal tubular lipidosis, and 2 others (1 male and 1 female) had similar hepatic and renal lipidosis. The remaining two otters in this group were both male; no significant lesions were found.

The six apparently healthy sea otters collected from an area that had not been affected by an oil spill (Group No. 4) did not have interstitial pulmonary emphysema, gastric erosions, hepatic or renal lipidosis, hepatic necrosis, or other significant lesions. Four were male and two were nonpregnant females.

Interstitial pulmonary emphysema was prevalent in oil-contaminated sea otters that died in rehabilitation centers (19/46). The lesion was more frequent in the more extensively contaminated otters. It was also present in 1/5 otters found dead at the site of the spill with external oil present. Emphysema was not seen in uncontaminated otters that died in rehabilitation centers nor in apparently normal otters. The pathogenesis of the lesion in this setting is unclear. Alveolar tears are the usual route by which air enters the pulmonary interstitium. Alveolar tears can occur when there is a combination of forced expiration or coughing and bronchiolar obstruction that produces sharply increased pressures within alveoli. In anatomically predisposed species such as cattle, the lesion may occur agonally, presumably due to forced expiration combined with bronchiolar collapse.

Predisposing factors include well-developed interlobular septa, lack of collateral ventilation, and greatly uneven deflation among adjacent lobules. Sea otters have well-developed interlobular septa and thus, may be anatomically predisposed to development of interstitial emphysema, but exposure to crude oil resulted in a remarkably high incidence of the lesion. During the early days of the spill, inhalation of volatile components of crude oil such as benzene might have damaged alveolar septa and caused the lesion, but neither interstitial pneumonia nor other lesions that might result from inhalation of an irritant vapor were found in affected sea otters. Thus, the pathogenesis of interstitial pulmonary emphysema in oil-contaminated sea otters is undetermined.

Gastric erosions were common (13/40) in oil-exposed sea otters that died in rehabilitation centers and were also found in one of the six uncontaminated otters that died in the centers. Melena was reported in many otters in the rehabilitation centers. Rapidly developing gastric erosions that appear following severe stress occur in humans beings and animals.

Gastrointestinal erosion/ulceration and hemorrhage have been reported in sea otters that died in captivity and in the wild and have been attributed to stress. All of the gastric erosions seen in this study were acute; none showed signs of healing. Those present in otters that died shortly after arrival at the rehabilitation centers might have developed prior to capture because of stress associated with oil exposure, as a direct effect of oil on the gastric mucosa, or because of stress associated with capture and captivity. Erosions caused by ingestion of corrosive liquids are extensive, but the erosions we

encountered were small, discrete, and confined to the stomach. Thus, we believe the erosions were probably caused by stress.

Hepatic lipidosis was common in oil-contaminated otters that died in rehabilitation centers (14/47) and in oil-contaminated otters that were found dead (3/5). It was also seen in an uncontaminated otter that died in a rehabilitation center (1/6). Renal lipidosis was somewhat less common and occurred only in otters that also had hepatic lipidosis. Hepatic and renal lipidosis have various causes including toxins, mobilization of stored fats due to inadequate food intake, and hypoxia. Hepatic and renal lipidosis may have been caused by an oil exposure-associated increase in energy demand with constant or decreased food intake resulting in mobilization of stored fat. A direct or metabolite-associated toxic effect is another possible cause.

Centrilobular hepatic necrosis was also relatively common in oil-contaminated otters that died in rehabilitation centers (11/47) and was not found in uncontaminated otters that died in the centers. Causes of centrilobular hepatic necrosis include toxins and conditions that cause hepatic ischemia, such as anemia, heart failure, and shock.

Some oil-contaminated otters became anemic while at rehabilitation centers; this may have contributed to the development of centrilobular necrosis. Crude oil ingestion and gastric erosion and hemorrhage are possible causes of anemia; however, gastric erosions and centrilobular hepatic necrosis were found infrequently in the same otter, so anemia due to gastric hemorrhage was not a common cause of centrilobular hepatic necrosis. Other lesions likely to occur in heart failure were not found. It is prob-

able that many otters experienced shock. Various incidental lesions were found in animals in all groups.

Sea otters are largely dependent on the insulating properties of their pelage for protection from the cold water they inhabit. It had been suspected that hypothermia would be a major problem in oil-contaminated sea otters because oil markedly increases the thermal conductance of their coats, and indeed hypothermia was a common problem in oil-contaminated sea otters presented to rehabilitation centers. Death caused by hypothermia can occur without distinctive gross

or microscopic lesions. It is likely that stress and shock were significant medical problems. Both oil exposure and captivity are stressful to sea otters. We believe that hypothermia, stress, shock, respiratory compromise associated with interstitial emphysema, hemorrhage from gastric erosions, and hepatic necrosis contributed to the deaths of oil-exposed sea otters.

Note: This extended abstract is excerpted from the following manuscript: Lipscomb, T. P., R. K. Harris, R. B. Moeller, J. M. Pletcher, R. J. Haebler, B. E. Ballachey. 1993. Histopathologic lesions in sea otters exposed to crude oil. *Vet Pathol* In press.



Gross Lesions in Sea Otters Found Dead Following the *Exxon Valdez* Oil Spill

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Following the *Exxon Valdez* oil spill in Prince William Sound, Alaska, sea otter carcasses were collected from areas affected by the spill, placed in plastic bags, and frozen. Beginning 15 months later and continuing over a 3 month period, 214 of the carcasses were thawed and autopsies were performed. Specimens were collected for toxicological testing for petroleum hydrocarbons. Because of the artifacts produced by freezing and thawing of tissues, histologic examinations were not performed.

Of 152 sea otters with external oil present, 100 (66%) had interstitial pulmonary emphysema and 83 (55%) had gastric erosion and hemorrhage; 64 (42%) oil-contaminated otters had both of these lesions. Of 62 sea otters with no external oil found, 13 (21%) had interstitial pulmonary emphysema and 4 (6.5%) had gastric erosion and hemorrhage; all 4 (6.5%) that had gastric erosion and hemorrhage also had interstitial pulmonary emphysema. Among sea otters with no detectable external oil, 2 had lesions likely to have contributed to their deaths: 1 had vegetative valvular endocarditis and the other had a gun shot wound in the thorax. Both of these sea otters also had interstitial pulmonary emphysema. A variety of incidental lesions were found in many of the sea otters.

A study of histopathologic lesions in oil-contaminated sea otters that died in rehabilitation centers following the *Exxon Valdez* oil spill found interstitial pulmonary emphysema in 19/46 (41%) and

gastric erosion and hemorrhage in 13/40 (32.5%). Among six uncontaminated sea otters that died in rehabilitation centers, interstitial pulmonary emphysema was not found and gastric erosion and hemorrhage were found in one¹. Based on the findings in the present study and the histopathologic study, we conclude that interstitial pulmonary emphysema and gastric erosion and hemorrhage are associated with crude oil exposure in sea otters. (Hepatic and renal lipidosis and hepatic necrosis were also relatively common histopathologic lesions in oil-contaminated sea otters; these lesions could not be specifically identified by gross examination alone).

We considered those sea otter carcasses that were oil-contaminated, had one or both gross lesions associated with crude oil exposure (interstitial pulmonary emphysema and/or gastric erosion and hemorrhage), and that did not have lesions indicative of another possible cause of death, as having strong evidence of death due to crude oil exposure. Sea otters fitting these criteria comprised 123/214 (57%). Those carcasses that were oil-contaminated and had neither lesions associated with crude oil exposure nor lesions indicative of another possible cause of death were considered to have evidence of death due to crude oil exposure. Sea otter carcasses that conformed to these criteria included 29/214 (14%). Those sea otter carcasses that had did not have detectable external oil, did not have lesions associated with oil exposure, and

did not have lesions indicative of another possible cause of death comprised 49/214 (23%) and were considered to have an undetermined cause of death.

Sea otter carcasses that did not have detectable external oil, did not have lesions indicative of another possible cause of death, and did have interstitial pulmonary emphysema and/or gastric erosion and hemorrhage included 11/214 (5%) were considered to have an undetermined cause of death. The uncontaminated sea otter that had a gunshot wound of the thorax and interstitial pulmonary emphysema and the uncontaminated sea otter that had vegetative valvular

endocarditis and interstitial pulmonary emphysema comprised 2/214 (1%) and were considered to have died primarily because of conditions unrelated to oil exposure.

Possible pathogeneses of lesions associated with crude oil exposure and mechanisms by which crude oil exposure might result in deaths of sea otters are discussed elsewhere (Lipscomb, 1993). Results of toxicological studies are not yet available.

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Age Distributions of Sea Otters Dying in Prince William Sound, Alaska, After the *Exxon Valdez* Oil Spill

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Sea otters (*Enhydra lutris*) suffered severe injuries as a result of the *Exxon Valdez* oil spill. Acute losses, represented by 871 sea otter carcasses recovered during response activities and an additional 123 otters dying at rehabilitation centers, are well documented (Bayha and Kormendy, 1990; DeGange and Lensink, 1990). Additional number of otters almost certainly died within weeks of the oil spill, but their carcasses were not recovered. Estimates of acute loss have been provided by Bodkin and Udevitz (1993), Doroff et al. (1993) and Garrett et al. (1993).

Injuries to sea otters probably were not limited to the deaths associated with acute exposure to oil. Surviving otters were exposed to oil remaining in the environment. Specifically, elevated concentrations of hydrocarbons have been reported in sediments and tissue samples of important sea otter prey, providing a pathway for continued exposure to sea otters. Little is known, however, about possible long-term sub-lethal effects on sea otters of initial exposure to oil or continued exposure to residual oil through the environment or prey. Because fresh carcasses of dead otters with known oil exposure histories are rarely collected, we must look for effects at the level of the population.

Bodkin and Jameson (1991) concluded that carcass recovery rates were not useful in estimating mortality rates in sea otters. However, systematic surveys for

beach-cast carcasses can provide useful information on temporal patterns of mortality and on the age class structure of the dying portion of the sea otter population under study (Kenyon, 1969; Bodkin and Jameson, 1991). A general pattern of the age class structure of sea otters dying in a population experiencing natural mortality has been described in California (Bodkin and Jameson, 1991), Amchitka, Alaska (Kenyon, 1969) and Prince William Sound (Johnson, 1987). The observed pattern includes a bimodal age distribution with the first mode consisting of young animals and the second mode consisting of aged animals. The proportion of prime-age (2-8 years) sea otters is relatively low.

A description of the age distribution of sea otters dying in western Prince William Sound in 1990 and 1991 thus allows a comparison against an expected mortality pattern. A change in the age class structure from the general pattern described above may indicate abnormal mortality representative of continuing deleterious effects of oil exposure. In this study, we examine patterns of sea otter mortality following the *Exxon Valdez* oil spill by estimating the ages of sea otters recovered as carcasses in western Prince William Sound, and comparing age distribution among pre-spill, spill and post-spill years.

During 1974 to 1984 (Johnson, 1987), and again in 1990 and 1991, sea otter carcasses were recovered during system-

atic beach surveys in the late spring in the area of Green Island in western Prince William Sound. Systematic surveys were not done in 1989, the year of the oil spill, but 493 sea otter carcasses were recovered from western Prince William Sound during response efforts. Sea otter carcasses were also collected in spring and summer, 1990 and 1991, from areas in the western Sound (i.e., in addition to Green Island) during oil spill clean-up and monitoring activities.

A premolar tooth was collected from the skull (when present) of each carcass and analyzed for age, based on cementum annuli (Garshelis, 1984). Carcasses were classed as juvenile (0 or 1 year of age), prime-age (2 to 8 years of age) or aged (9 years or older). Comparisons were made among age class distributions of sea otter carcasses recovered from 1974 through 1984 (pre-spill), in 1989 following the oil spill (spill year), and in 1990 and 1991 (post-spill). Differences among distributions were tested using Fisher's Exact Test (2 tailed).

Age was determined for 142 sea otter carcasses recovered at Green Island pre-spill. Of these, 45% were from juveniles, 15% from prime-age and 40% from aged adult otters (Johnson, 1987). Age was determined for 437 carcasses recovered in western Prince William Sound in the spill year; 32% were from juveniles, 44% from prime-age and 24% from aged otters. Carcasses recovered at Green Island and in other areas of western Prince William Sound in 1990 and 1991 were combined: age was determined for 66 individuals, with 33%, 43% and 24% of the carcasses from juvenile, prime-age and aged otters, respectively. The proportion of prime-age carcasses recovered increased in both the spill year and in post-spill years, relative to the pre-spill

years. Differences between pre-spill and spill year age distributions, and between pre- and post-spill age distributions were significant ($P < .0001$ and $P = .0005$, respectively). Spill year and post-spill years did not differ significantly.

The age distribution of sea otters dying in western Prince William Sound since the spill differs from the generalized pattern previously described for sea otters (Bodkin and Jameson, 1991; Kenyon, 1969; Johnson, 1987). The observed differences represent a shift in the age-class composition of sea otters dying in western Prince William Sound from a pre-spill, bimodal distribution, composed of primarily young and old animals, to an increased proportion of prime-age animals in years following the oil spill. This shift suggests that higher than normal proportions of prime-age sea otters are dying in some portions of the Sound, and may be evidence of a prolonged, spill-related effect on the sea otter population affected by the *Exxon Valdez* oil spill.

Several assumptions are made in reaching this conclusion: (1) carcasses recovered in a given geographic area are of sea otters that lived in that general area prior to their death, (2) carcasses recovered in 1989 or in 1990 and 1991 represent deaths of the year in which they were recovered, (3) mortality of sea otters in 1989 associated with the oil spill was non-selective with regard to age, and (4) carcasses of all dying sea otters have an equal probability of being recovered, regardless of age of the otter. We believe that the first three assumptions are largely met. The last assumption may be violated to a greater extent: carcasses of juvenile sea otters may have a lower probability of recovery than carcasses of adult otters. If such a bias exists,

however, it should have affected the pre-spill, spill year and post-spill age distributions equally, and comparisons across years are valid. We conclude that the abnormal post-spill mortality pattern for sea otters, based on age distributions of dead animals, provides strong evidence of continuing injury to sea otters in western Prince William Sound through at least 1991.

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Recovery of Sea Otter Carcasses Following the Exxon Valdez Oil Spill

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In response to the T/V *Exxon Valdez* oil spill of 1989, sea otter (*Enhydra lutris*) carcasses were recovered in or adjacent to the oil spill area. Carcasses were collected between March 30 and September 15 and brought to collection facilities at Valdez, Seward, Homer, or Kodiak (DeGange and Lensink, 1990). Carcasses were collected and transported by federal and state response staff as well as private contractors and volunteers. We report on the distribution, degree of oiling and state of decomposition, age and sex structure, and reproductive status of recovered carcasses. We discuss factors influencing the recovery of carcasses. We also provide an estimate of acute sea otter mortality as a result of the T/V *Exxon Valdez* oil spill based on the probability of recovery of carcasses and compare our estimate with those generated by other methods.

Carcasses in suitable condition were subjectively scored for relative degree of external oiling and extent of decomposition. Weight, length, general age class (adult, subadult or pup) and reproductive condition (including presence or absence of lactation) were recorded. Reproductive tracts were removed from adult females and later examined for reproductive history (USFWS, unpublished data). Skulls and baculums (when present) were removed from all carcasses. Complete necropsies were performed on 214 suitable carcasses (Lipscomb et al., in press). The first premolar tooth was taken for age determination (Garshelis, 1983), and tissue samples were collected

for toxicology (Mulcahy and Ballachey, 1993). Data on recovered carcasses were often incomplete due to a lack of preparation for an event of this magnitude.

A total of 871 sea otter carcasses were collected during response activities, and 123 live otters were captured and subsequently died at rehabilitation centers. Ninety of the recovered carcasses were judged to have died prior to the spill, and are not considered in the following analyses. One hundred and twenty of the carcasses were not evaluated as to time of death relative to the spill, and are included in the analyses. The remainder (904) reflects a minimum estimate of acute sea otter mortality. It is almost certain that additional sea otters were oiled and subsequently died, and their carcasses were not recovered. In addition, live oiled otters that would have died and become carcasses, but were captured for cleaning, became unavailable for recovery as carcasses. The 123 otters that died at rehabilitation centers are not considered in the following analyses. However, the number of potential carcasses may have been reduced by the extent that the rehabilitation effort reduced mortality. The 123 otters that died at rehabilitation centers are considered only in the loss estimate.

From Prince William Sound 424 carcasses were recovered, 167 from the Kenai Peninsula, and 190 from the Kodiak Island/Alaska Peninsula area. The proportions of otters which apparently died before the spill were estimated at 15%, 9% and 5% for Prince William Sound,

Kenai Peninsula and Kodiak Island/Alaska Peninsula, respectively.

The proportion of oiled carcasses and the degree of oiling (heavy, moderate, light or none) generally declined as the distance from the spill increased. Percentages of oiled carcasses for the three areas were 93%, 68% and 52%, for Prince William Sound, Kenai Peninsula and Kodiak/Alaska Peninsula, respectively. Heavily and moderately oiled carcasses were 67%, 48% and 30% from the three areas, respectively. The proportion of lightly oiled carcasses was relatively stable at 26%, 20% and 22%, respectively. Non-oiled carcasses were 7%, 32%, and 48% from the three areas, respectively.

The condition of carcasses was similar among the three recovery areas. Seventy-three percent, 66% and 63% of the carcasses were considered relatively fresh with organs discernable and skeleton intact from Prince William Sound, Kenai Peninsula and Kodiak/Alaska Peninsula, respectively.

We estimated ages of 670 of the carcasses recovered: 400 from Prince William Sound and 135 each from the Kenai Peninsula and Kodiak/Alaska Peninsula. Three age classes were defined: adults (3 years and older), subadults (1 and 2 years) and pups (0 years). Adults comprised 58%, 57% and 33%, subadults 33%, 13% and 10%, and pups 9%, 30% and 57% of the carcasses from Prince William Sound, Kenai Peninsula and Kodiak/Alaska Peninsula, respectively.

Of the adults, the proportion of females was 67% and 65% from Prince William Sound and the Kenai Peninsula, respectively. From the Kodiak/Alaska Peninsula region, the numbers of adult males and females recovered were approximately equal (48% female); however, sex was not determined for almost

half the carcasses. We found 62% of the female carcasses from Prince William Sound pregnant and 15% lactating. At Kenai, 30% were pregnant and 33% lactating. This comparison may reflect differences in the reproductive phenology between these two areas.

Two experiments were conducted following the spill to evaluate the carcass recovery efforts (Doroff and DeGange, 1992). One study monitored the movements of radio-equipped simulated sea otter carcasses to identify patterns of movements and fate. A pertinent result of this study was that a small proportion (2 of 30) of the simulated carcasses left Prince William Sound.

The second study was designed to assess the probability of carcass recovery. Twenty-five previously recovered then marked sea otter carcasses were released in May and June 1989 near northern Kodiak Island in an area where wildlife rescue and beach clean-up crews were operating. The carcasses varied in size, degree of decomposition, and whether they were frozen or thawed at the time of release. Five of the 25 carcasses (20%) were recovered. Similar recovery rates have been obtained from similar studies in California.

Assuming the results of this experiment are applicable to the overall spill area, and excluding those carcasses which were determined to have died before the oil spill, the loss estimate associated with the oil spill would be 4,028 (781 / .20, plus 123 dead at rehabilitation centers) sea otters. Assumptions made in estimating this loss are: (1) recovery rates estimated at Kodiak in May and June are representative of the entire oil spill area and time period during which carcass collection occurred, (2) the carcasses used in the experiment did not differ in recoverabil-

ity from fresh carcasses of sea otters dying from oil exposure, and (3) the 123 otter deaths at the rehabilitation centers are oil related. The degree to which these assumptions are not met and their effects on the accuracy of loss estimates are unknown. This loss estimate thus should be used conservatively.

Two other methods were used to estimate acute mortality to sea otters. In one, estimates of sea otter abundance pre and post-oil spill were used to generate a loss estimate of 2,787 sea otters for Prince William Sound (Irons et al., 1988; Burn, 1992; Garrett et al., in prep). Boot strapping techniques were used to generate confidence limits of 500-5,000 sea otters on the loss estimate. Using our carcass recovery rate of .20 we estimate an acute loss of 2,209 sea otters in Prince William Sound (424/.20 plus 89 from the rehabilitation centers).

The third approach involves a predictive model of mortality based on the potential exposure of otters to oil and observed mortality rates associated with different levels of exposure. Initial results of this model indicate 497 sea otters may have been exposed to lethal levels of oiling on the Kenai Peninsula (Bodkin and Udevitz, in press). Using our carcass recovery rate of .20 we estimate an acute loss of 868 from the Kenai Peninsula (167/.20 plus 33 from the rehabilitation centers).

Of these three approaches, only the estimate based on a carcass recovery rate provides a loss estimate for the entire spill area. Limitations on pre-spill population data (distribution and abundance) restrict applications of the second approach. The third approach, with further refinement, may be extended to oiled areas other than the Kenai Peninsula. It

should be noted that current estimates have not accounted for the loss of reproductive potential from the sea otter population associated with females pregnant at the time of death.

After the oil spill, we were left to reconstruct pre and post-spill sea otter populations with limited data. Loss estimates were generated with wide confidence limits and incorporated numerous, untestable assumptions. In retrospect, additional studies implemented at the time of the oil spill may have improved the quality of the population loss estimates. For example, a more thorough study of marked and released carcasses and their subsequent recovery implemented on a broad geographic scale may have helped refine estimates of carcass recovery probability and thereby our ability to estimate mortality.

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An Intersection Model for Estimating Sea Otter Mortality Following the Exxon Valdez Oil Spill

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There are several possible approaches for estimating the number of sea otter mortalities resulting from acute exposure to Exxon Valdez oil. One approach is to apply an estimate of carcass recovery probability to the number of carcasses actually recovered to obtain a total acute loss estimate (Doroff et al. 1993). However, data on recovery rates were limited in geographical scale and sample size, and the applicability of these rates to all areas in the spill is questionable.

A second approach is to take the difference between sea otter population sizes estimated before and after the spill (Burn 1990, DeGange et al. 1990, Douglas et al. 1990). Incomplete coverage by pre-spill surveys prevented direct application of this approach to all areas affected by oil. For the Kenai Peninsula, in particular, pre and post-spill data did not indicate a statistically significant loss and carcass recovery rates were not available. Therefore, we investigated a third approach that uses an analytical model (intersection model) to estimate exposure to oil and subsequent mortality of sea otters along the Kenai Peninsula.

The intersection model relied on three distinct data sets from the Exxon Valdez oil spill. The first was the distribution and movement of spilled oil as it traveled throughout the spill area as modeled by Galt et al. (1991). The second data set resulted from a helicopter survey of sea otter abundance and distribution conducted concurrently with the movement of oil along the Kenai Peninsula (DeGange

et al. 1990). The third data set consisted of the number of sea otter carcasses that were recovered (U.S. Fish and Wildlife Service, unpub. data) combined with the number of otters that were captured alive at two sites in Prince William Sound that received two different exposure levels (light and heavy) to oil (Bodkin and Wetz 1990). The subsequent survival of live otters captured and the number of carcasses recovered were used to estimate mortality rates at these two exposure levels.

The National Oceanic and Atmospheric Administration On-Site Spill Model (OSSM) traced the movement of 10,000 lagrangian elements (oil particles), each representing about 1,100 gallons of oil, at three hour intervals from 24 March through 29 April, 1989. The volume of oil represented by each element remained constant, while the number of particles declined as a result of evaporation, sinking or beach-cast oil.

We delineated an exposure area consisting of a circle (buffer) 2.8 km in diameter at each location where sea otters were observed during the helicopter survey. This diameter represents twice the average distance sea otters were observed to move between successive radio locations recorded between 18 and 36 hours apart in California (Ralls et al. 1988). The exact size of the buffer should not be critical because our model only makes use of the rank ordering of exposure levels and therefore should be sensitive only to very large changes in buffer size.

The exposure level at each location was estimated by counting the number of oil particles (OSSM) that intersected the associated buffer at least once during each 24-hour period following the spill (gallon*day). Then the exposure levels were extrapolated to the population based on the otter group size at each observed location and the population size estimate of Douglas et al. (1990).

The result was a relative measure of the quantity and duration of oil present in the vicinity of each otter in the Kenai Peninsula population. The same procedure was conducted for random locations within the two capture sites in Prince William Sound. We then applied the known mortality rate estimates associated with exposure levels at the Prince William Sound capture sites to the otter population along the Kenai Peninsula.

The intersection model indicated that 131 sea otter buffers were intersected by one or more oil particles during the course of the spill. This corresponds to an estimated 1,211 sea otters that were exposed to oil along the Kenai Peninsula. Modeled exposure levels along the Kenai (1-141 gallon*days) were most similar to the "light oiling" capture site in Prince William Sound (66-438 gallon*days) where the mean estimated mortality rate was 0.39 (n=33).

Simply applying this rate to all exposed sea otters along the Kenai, resulted in a loss estimate of 472 individuals. An alternative method assigns a mortality rate of 0.20 to animals exposed to 1 to 66 gallon*days and a mortality rate of 0.39 to otters with higher exposure, resulting in a loss estimate of 284 otters. Alternative loss estimates may be obtained by using additional information or altering assumptions about the relationship between exposure and mortality.

Several assumptions were required for the use of the data sets employed by the intersection model in addition to the assumptions of the model itself. The most important of the data assumptions are: (1) that the OSSM model accurately reflects the spatial and temporal distribution of spilled oil, (2) that estimates of sea otter distribution and abundance obtained from the helicopter survey were unbiased, (3) that sea otter collection methods at the two capture sites were not biased, and (4) that mortality rates for captured otters were not affected by the capture or the rehabilitation process. The three key assumptions of the model are (1) that mortality was in fact, a function of exposure to oil, as measured by gallon*days, (2) the relative exposure level measured for the region around each observed otter location reflects the relative exposure of those otters to oil, and (3) the relationship between exposure and mortality at the capture sites in Prince William Sound was the same as the relationship at the Kenai Peninsula. The likelihood that these assumptions are valid and their potential effects on estimated loss are discussed.

The intersection approach appears to provide an adequate estimate of relative exposure to oil, in cases where there are suitable data on abundance and distribution of sea otters. Because loss estimates are sensitive to the relationship between exposure to oil and mortality rates, refinement of the model should include a better definition of this relationship, particularly at low exposure levels. However, since the rehabilitation treatment is likely to effect survival, it is doubtful that otters subjected to rehabilitation will provide useful information about the relationship of exposure and mortality in otters that are not subjected

to rehabilitation.

Accurate and precise surveys of sea otter abundance are required to provide reliable loss estimates based on comparisons of pre and post-spill surveys. However the intersection approach also relies, in part, on accurate and precise surveys of the population. Improved survey methods, implemented on a routine basis in areas of oil storage and transportation would provide direct estimates of loss in the event of future oil spills and eliminate the need for indirect approaches such as the intersection model.

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Mortality of Sea Otters in Prince William Sound Following the Exxon Valdez Oil Spill

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This paper presents an estimate of the total number of sea otters that died as a direct consequence of the oil spill that occurred when the *Exxon Valdez* grounded in Prince William Sound, Alaska, on March 24, 1989. We compared sea otter counts conducted from small boats throughout the Sound during the summers of 1984 and 1985 to counts made after the spill during the summer of 1988. We used ratio estimates, corrected for sighting probability, to calculate otter densities and population estimates for portions of the Sound affected by the oil spill.

We estimated the otter population in the portion of Prince William Sound affected by the oil was 6,685 at the time of the spill and that the post-spill popula-

tion in the summer of 1989 was 3,898, yielding a loss estimate of approximately 2,800. Bootstrapping techniques were used to approximate confidence limits on the loss estimate of about 500-5,000 otters. The wide confidence limits are a result of the complex scheme required to estimate losses and limitations of the data.

Despite the uncertainty of the loss estimate, it is clear that a significant fraction of the otters in the spill zone survived. We observed otters persisting in relatively clean embayments throughout the oil spill zone, suggesting that the highly convoluted coastline of Prince William Sound produced refuges that allowed some sea otters in the oil spill area to survive.

Hydrocarbon Concentrations In Tissues Of Sea Otters Collected Following The Exxon Valdez Oil Spill

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The concentrations of 28 aliphatic and 39 aromatic hydrocarbons were measured in up to seven tissues (liver, muscle, kidney, brain, intestine, fat, and testes) from ten sea otters recovered dead in western Prince William Sound between April 5 and 11, 1989.

All animals were subjectively judged to be heavily or moderately oiled. Six animals were males and four were females, two of which were pregnant when they died. Ages ranged from 1 to 9 years old and weights from 8.3 to 22.8 kg. Data were transformed using a $\log_{10}(i+1)$ formula, where i was the raw hydrocarbon concentration. Data on hydrocarbons from liver, muscle and kidney from ten sea otters killed as part of a subsistence hunt in southeast Alaska were used as controls unexposed to the T/V Exxon Valdez oil spill.

Concentrations of aliphatic hydrocarbons ranged between 10 and 1000 ng/g. With the exception of fat tissue, concentrations and prevalences of the normal alkane hydrocarbons in the range from n-C10 to n-C15 and n-C30 to n-C34 were lower than the n-C16 to n-C29 alkanes. Fat tissue frequently had higher concentrations of hydrocarbons than other tissues.

The aromatic hydrocarbons generally had concentrations of less than 100 ng/g of tissue except for fat and intestine, which had higher overall concentrations. The highest prevalences and concentrations of aromatic hydrocarbons

in all tissues occurred with naphthalene, alkylated naphthalene, and biphenyl. Every sample from all of the sea otters had some concentration of naphthalene and C1-naphthalene. Phenanthrene was present at low concentrations in all tissues except muscle. In addition, dibenzothiophene and alkylated dibenzothiophene at low concentrations occurred in greater frequency in fat, brain and intestine than in the other tissues. Fat had the highest overall concentrations of aromatic hydrocarbons.

Exposure of the sea otters to crude oil was most clearly evidenced by the elevated concentrations of a wide range of aliphatic and aromatic hydrocarbons in at least some tissues. In addition, the presence of the unresolved complex mixture fraction and the presence and elevated concentration of alkylated derivatives of several aromatic compounds were important variables in determining exposure to crude oil.

Based on prevalences and concentrations of hydrocarbons in the tissues of the animals examined, a pattern of exposure and mortality was proposed for each sea otter. Two of the sea otters had very low concentrations of hydrocarbons in their tissues, suggesting a very limited exposure to or movement of hydrocarbons into internal organs prior to death. Hydrocarbon patterns in intestine samples indicated that three of the ten animals clearly had ingested oil. The remaining animals had intermediate pat-

terns of hydrocarbons in their tissues.

We propose that sea otter VD074 ingested crude oil, but died before the hydrocarbons could be distributed to the other tissues to any great degree. This sea otter had extremely high concentrations of alkane hydrocarbons in the sample of intestine, which was the only sample in the study that contained detectable concentrations of every alkane hydrocarbon analyzed.

However, concentrations of alkane hydrocarbons in the other tissues from this animal were not elevated. The unresolved complex mixture fraction was found only in the intestine and was present at the highest concentration (905 ng/g) of any of the samples in this study.

The concentration of the total of all aromatic hydrocarbons was also very high in the intestine; alkylated derivatives of naphthalene, phenanthrene, fluorene, dibenzothiophene and chrysene were present at higher concentrations than their parent compounds. Concentrations of these compounds were also elevated in other tissues from this sea otter compared to the same tissues from other sea otters.

Sea otter VD141 is an example of an animal with very low concentrations of hydrocarbons in its tissues despite heavy external oiling. The low concentrations and prevalences of hydrocarbons suggest that this animal died quickly following exposure to oil and had insufficient time before death to move hydrocarbons into its tissues. The unresolved complex mixture fraction was not present in any tissue. The concentrations of alkane hydrocarbons across the range of compounds were similar between tissues, and a pattern of slightly lower concentrations was seen in intestine and muscle than in the other tissues.

This animal had the lowest overall concentration of total aromatic hydrocarbons of the sea otters studied. The parent compounds naphthalene, phenanthrene, fluorene, dibenzothiophene, and chrysene were found at low concentrations, but, of their alkylated derivatives, only C1-naphthalene was detected in tissues.

The overall concentrations of hydrocarbons measured in tissue samples from sea otters found dead and grossly contaminated with crude oil were very low, ranging from 0 to <1,000 ng/g. The same individual hydrocarbons and many of the same sums and ratios of hydrocarbons that are used in studies of crude oil and oiled sediments are applied to studies of oiled marine mammals (Hall and Coon, 1988).

Very little is known about the fate of hydrocarbons in marine mammals. In common with other higher animals, marine mammals have inducible enzyme systems permitting the metabolism and excretion of systemic hydrocarbons (Engelhardt, 1982; Addison and Brodie, 1984; Addison et al., 1986). The actions of these enzymes may account for the low concentrations in tissues of hydrocarbons found in crude oil.

The large number of variables (hydrocarbons) for each sample, the extremely wide range of concentrations, and the non-normal distribution of the raw data complicated analysis. We are presently using multivariate statistics for reduction and analysis of the data set as has been used recently by others analyzing contaminants data (Cripps, 1992; Devillers and Karcher, 1991; Finzi et al., 1991; Georgakopoulos-Gregoriades, et al., 1991; Saenz and Pingitore, 1991).

The sea otters in our study were dead when collected. Other than collection

site and date, no information was known about the type and duration of exposure of the animals to crude oil. Although all of the sea otters were collected soon after the oil spill occurred and the carcasses were in good condition and were coated with moderate to heavy amounts of oil, no factual information is known about how long they lived following exposure. Some may have died from causes other than oiling and have suffered postmortem oiling.

These considerations, in the face of potential litigation, point out the importance of analyzing for tissue hydrocarbon residues even in animals with obvious external oiling, and the need for careful interpretation of the data.

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Health, Reproduction, and Survival of Adult, Dependent and Weanling Sea Otters in Prince William Sound From October 1989 to December 1991.

Lisa Mignon Rotterman and Charles Monnett
Enhydra Research

In order to determine the damage to the sea otter population resulting from the T/V *Exxon Valdez* oil spill, we obtained data on the health, behavior, reproduction, and survival of several crucial demographic classes of sea otters (*Enhydra lutris*) in areas within, and adjacent to, the oil spill zone. These were: adult females, male and female dependent pups (pups still with their mothers), and male and female weanlings (individuals caught while with, and dependent upon, their mothers, but who became independent over the course of the study). We also studied a small sample of adult males, but will not report on those here.

Thus, ninety-six adult female (eastern Prince Sound: $n = 44$; western Prince William Sound, the oil spill area: $n = 51$), and sixty-four dependent (eastern Prince William Sound: $n = 24$; western Prince William Sound: $n = 40$) sea otters were captured, examined, instrumented with radio-transmitters, and monitored between October 1989 and November 30, 1991.

Individuals in the oil spill region were compared to individuals of the same age and sex in eastern Prince William Sound.

Comparable data were also available from 104 sea otters captured and instrumented in Prince William Sound in previous years.

Preliminary examination of several indices indicate that sea otters in the oil spill area were in relatively poor health compared to their counterparts in non-oiled areas of Prince William Sound.

Additionally, preliminary data indicate that survival rates may vary between sea otters in oiled and unoled portions of the Sound. These data are discussed and interpreted with regard to the following issues: overall extent of damage to the sea otter population(s); estimation of the rate and pattern of recovery of the population studied; identification of critical habitat areas for sea otters in Prince William Sound; determination of the relative quality of different habitats; and formulation of current and future restoration and response activities.

Preliminary evaluation of data as a whole indicate that it is likely, at the time our monitoring of individuals in this study ended, the population of sea otters in western Prince William Sound was not yet in a recovery phase.

Effects of the Exxon Valdez Oil Spill on River Otters in Prince William Sound

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Effects of the Exxon Valdez oil spill on river otters (*Lutra canadensis*) inhabiting Prince William Sound, Alaska were investigated from 1989-1992. These river otters feed mostly on marine invertebrates and fishes, which they obtain by foraging primarily in subtidal and intertidal zones in the Sound. Consequently, we expected these mustelids to be strongly affected by the oil spill. Indeed, river otters population elsewhere have provided a sensitive indicator to aquatic pollutants.

Initially, our research efforts were focused on two 80-km areas: Northern Knight Island, including Herring Bay (which received heavy oiling), and Esther Passage (which was outside the path of the spill). These areas were selected because they were ecologically similar and, based on number of latrine sites, had similar densities of otters. This design was selected over one that examined otters on oiled and nonoiled areas within the path of the spill, because a mammal with the mobility of an otter undoubtedly would have been exposed to areas polluted by oil within its home range, thereby biasing our data.

Some carcasses of river otters were recovered from beaches immediately following the spill and, for specimens that were not too badly decomposed, necropsy and hydrocarbon analyses implicated exposure to crude oil in their deaths. Assessment of acute mortality to river otters resulting from the oil spill,

however, is problematic. Sick and dying otters likely moved to dens or other areas above the shoreline and thereby avoided discovery by oil-spill workers. This assessment is further complicated by the absence of pre-spill data on the density of otters.

Nonetheless, we used a combination of radio-telemetry and isotope implants to calculate mark-recapture population estimates for otters on oiled (Northern Knight Island) and nonoiled (Esther Passage) areas in 1990. Confidence limits for our population estimates overlapped for most surveys.

Care must be used, however, in interpreting this outcome because we lack pre-spill estimates. Moreover, the method we used is most likely to detect large changes in population size, but not small or subtle ones. Provided that latrine sites were a reliable index to river otter populations, it is probably safe to conclude that catastrophic mortality of otters did not result from acute effects of exposure to crude oil. This methodology does not, however, rule out chronic mortality caused by the oil spill.

To examine the chronic effects of oil exposure on river otters, we proposed assaying specific blood parameters that would reflect trauma or toxic effects from oil. This was possible because we had drawn blood samples when otters were radio-implanted. River otters from oiled areas exhibited significantly ($P < 0.05$) higher levels of haptoglobin, indicative

of physiological damage, than otters from unoiled zones. Although other causes of elevated haptoglobins are possible, we know of no other single factor except oil that was likely to affect otters along the 80 km of shoreline where they were sampled.

Likewise, these same river otters exhibited significantly lower body mass on oiled compared to nonoiled areas. River otters might be exposed to oil by grooming it from their fur, eating oil-covered prey, or ingesting food (e.g., mussels) that had concentrated hydrocarbons. Elevated blood values and reduced body mass for otters on oiled sites might result from internal damage from oil, reduced digestibility of oiled food, a reduction in prey, or some combination of these factors.

We also studied the food habits, movements, and habitat selection by river otters. Because we collected the feces of river otters from latrines immediately following the spill, we have an index to the pre-spill diet of these mustelids. Analysis of feces for food items, however, cannot be used to reconstruct otter diets unless digestibility of each food is determined under laboratory conditions—an overwhelming task. Nonetheless, this technique (fecal analysis) does provide a valid index for making comparisons between areas (oiled vs nonoiled).

Based on this index, there was no difference in the diets of otters between oiled and nonoiled study areas in either late winter or summer 1989. Otter diets were highly diverse with bony fishes being the single most important item. A large change ($P < 0.05$) occurred in 1990, however, with about a 1/3 reduction in the number of food items occurring in otter scats on the oiled area, and with no

change on the nonoiled area. We cannot determine if this change was brought about by a reduction of prey on oiled areas, some change in their vulnerability to predation, or differences in otter behavior. Nevertheless, river otters from oiled areas obtained a less diverse diet than otters living in nonoiled areas.

River otters on oiled areas also had home ranges that were about twice as large ($P \leq 0.05$) as otters occupying nonoiled areas in 1990. Likewise, a comparison of active latrine sites with random sites along the coast indicated that river otters on oiled areas selected habitat differently ($P < 0.05$) than otters living in nonoiled areas.

Most notably, otters on oiled areas selected steep-sloped shores and avoided shallow ones, whereas otters on nonoiled areas did just the opposite. We interpret this to mean that otters on oiled areas avoided shallow-sloped sites, ostensibly because oiling was most severe and persisted the longest on shallower slopes. This interpretation is supported by otters on oiled areas of Herring Bay depositing feces at a rate that was lower ($P < 0.05$) than on unoiled sites within this same area.

A less diverse diet, lower body mass, larger home ranges, avoidance of preferred habitat, and elevation of blood parameters indicative of damage for river otters on oil areas is an overwhelming indication of the chronic effects of the oil spill. These effects occurred one year after the oil spill and following a major attempt to clean oil from contaminated beaches. It is likely that major effects of the oil spill on river otters were delayed, and that our population estimates were conducted too soon to reveal them.

One potential shortcoming of our data is that two 80-km areas may not be repre-

sentative of the whole Sound. Consequently, we live captured and drew blood samples from river otters from oiled and nonoiled areas throughout the Sound in 1991. This extensive study confirmed our results from intensive work on otter blood from Northern Knight Island and Esther Passage.

Additionally, we measured several new blood values (interleukin and SGOT) that suggest otters from oiled areas may be experiencing depressed immune systems. We also developed a biostatistical model that allowed us to predict ($P < 0.05$) whether otters came from oiled or unoled areas based on these blood parameters. We were able to continue examining the activities and habitat use by directly observing otters in Herring Bay and Esther Passage. Radio failures in Herring Bay, however, reduced sample size so that statistical comparisons are not meaningful. We were able to confirm, however, that latrine sites are centers of otter activity and reflect habitat use by otters.

Although changes in blood values indicative of exposure to oil were detected two years following the spill, the question as to how severely this affects river otter populations still remained. That some change in survivorship or recruitment of young should be expected can be inferred from blood values, food habits, home ranges and habitat selection differing on oiled and nonoiled areas. The timing and magnitude of such

change, however, remains uncertain.

To evaluate whether there was any overall effect on otters, we tested for rates of latrine site abandonment in oiled and nonoiled areas throughout the Sound. Latrine sites were abandoned at a rate that was over three times higher ($P < 0.05$) on oiled than nonoiled areas, raising the strong possibility that changes in populations of river otters were beginning in 1991.

In 1992, we again sampled river otter blood from animals live captured in oiled and nonoiled areas as part of a study on mussel beds. Analyses of these data are not yet complete, but at least one blood value is still elevated on oiled areas. There may be a declining trend, however, in the magnitude of these physiological responses with time.

Understanding the responses of river otters to oil contamination obviously requires pre-spill data and long-term monitoring. We detected differences in blood values of river otters that are related to their health three years after the spill. This raises important questions about the long-term effects of oil on river otters and other mammals similarly exposed to crude oil. Because of their position high on the food chain, river otters can serve as an "indicator species" for monitoring the health and recovery of animals at lower trophic levels in Prince William Sound.



Assessment of Damages to Harbor Seals Caused by the *Exxon Valdez* Oil Spill

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Some of the largest harbor seal haulouts in Prince William Sound and waters adjacent to these haulouts were directly impacted by substantial amounts of oil when the *Exxon Valdez* ran aground on Bligh Reef. Oil impacted seal habitat in the Gulf of Alaska at least as far to the southwest as Tugidak Island. A damage assessment study was conducted to investigate and quantify effects of the *Exxon Valdez* oil spill on the distribution, abundance, and health of harbor seals.

The study was conducted by the Alaska Department of Fish and Game, Division of Wildlife Conservation, in cooperation with the National Marine Fisheries Service, National Marine Mammal Laboratory. Almost all of the work was done in Prince William Sound because seals and their habitats were heavily oiled, and baseline information was available that could be used for comparison with data collected during and after the spill. The study included observations of seals in oiled and unoled areas, examination and sampling of carcasses of seals found dead and collected, and aerial surveys of trend count sites during pupping and molting.

In the weeks immediately following the spill harbor seals swam through oil and inhaled aromatic hydrocarbons as they breathed at the air/water interface. They did not appear to avoid oil in the water or on haulouts. In oiled areas, seals crawled through and rested on oiled rocks and algae throughout the spring and summer. During April-July 1989 we

saw no oiled seals in unoled areas that were not near or adjacent to oiled sites, but in oiled areas 50-100% of the seals usually were oiled. In May, over 80% of 585 seals observed in oiled areas were oiled, most of them heavily. Pups were born in May and June. In some cases both their haulouts and their mothers were covered with oil, and the pups became oiled shortly after birth.

Oiling of seals was most severe in central Prince William Sound (Smith Island, Little Smith Island, Seal Island and Applegate Rocks), the region from Eleanor Island through the north part of Knight Island (Northwest Bay, Upper and Lower passages, Bay of Isles, and Herring Bay), and the west side of Knight Island Passage (Crafton Island and Junction Island).

Oiling of seals was also documented from the Kenai Peninsula and the Barren Islands. Seals older than pups shed their oily coats during the August molt, and in 1990 we saw no sign of external oiling on any seals.

Harbor seals are known for being very wary and difficult to approach. However, in the weeks immediately following the spill oiled seals in oiled areas behaved very oddly, and were reported as being sick, lethargic, or unusually tame.

Seals often stayed on haulouts when aircraft flew over at low altitudes, and people were sometimes able to approach to within a few yards. During field work in 1990 and 1991, harbor seals were noticeably more wary and more difficult to

approach than they were in 1989.

We examined and necropsied 19 recently dead seals during April-July 1989. Fifteen were found dead, three died in captivity, and one was an animal shot by a subsistence hunter. Thirteen were pups, including two that died after about one month in rehabilitation facilities. Most of the carcasses were too decomposed to examine and sample properly. Two seals had broken bones and other injuries that suggested they had been hit by boats. Some dead pups appeared to have suffered from malnutrition and stress.

Twenty eight seals were collected for detailed examination and sampling. Twelve were collected in Prince William Sound during April-June 1989; all were oiled, most of them heavily. Six were collected from the Gulf of Alaska in June-July 1989; two were from Prince William Sound and the Gulf in October-November 1989; six were from Prince William Sound in April 1990; and two were from Ketchikan in August 1990.

With the exception of oily pelage on some of the animals, all the seals appeared normal based on external examination and measurements. Gross necropsy showed normal internal parasites. Conjunctivitis was found in six of the animals, all of which were collected from oiled areas.

Microscopic examination of tissues found debilitating lesions in the brains of many of the oiled seals collected. Lesions were most severe in the one animal collected in April 1989. Exposure to aromatic hydrocarbons caused swelling (intramyelinated edema) and degeneration of the nerve axons, which would have interfered with nerve transmissions. The lesions were mostly in the thalamus, a region of the brain that serves as a primary relay center for incoming and out-

going nerve impulses. This brain damage could have made it very difficult for seals to perform normal tasks such as swimming, diving, feeding, and escaping from predators. It would explain the unusual behavior by seals immediately after the spill, and probably made them more susceptible to drowning, being hit by boats, or being caught by predators such as killer whales. Volatile hydrocarbons are known to cause central nervous system damage in other mammals, and the intramyelinated edema found in oiled seals is similar to that present in humans that die from inhaling solvents. Damage that could have resulted from oil was also found in the eyes, skin, and liver of oiled seals.

In April-July 1989, values for hydrocarbon metabolites in bile were 7-13 times higher in seals collected from oiled parts of Prince William Sound than in those from the Gulf of Alaska. This confirms that seals took oil into their bodies through contact, inhalation, and/or ingestion. High levels persisted in the Sound at least through April 1990, which indicated that seals were still encountering oil in the environment, or that they were metabolizing stored fat reserves that had elevated levels of hydrocarbons.

Because seals have enzyme systems that allow them to detoxify and excrete hydrocarbons, the levels found in most tissues were not very high. In muscle the levels of total aromatic hydrocarbons were undetectable to very low (up to 10 ppb), while levels in brain (up to 64 ppb) and liver (up to 160 ppb) were somewhat higher. Highest levels generally occurred in the blubber of animals found dead and collected in Prince William Sound (up to 800 ppb). Relatively high levels were also found in mammary tissue (34-143 ppb) and milk (44-1200 ppb).

In 1989-1991, aerial surveys were flown along a trend count route that had previously been surveyed in 1983, 1984, and 1988. The route covered eastern and central Prince William Sound, and included seven oiled sites and 18 unoiled sites. Surveys showed that pup production was lower in oiled areas in 1989 than in 1990 or 1991, while in unoiled areas the ratio of pups to non-pups was similar in all three years. Together with the dead fetuses and pups found following the spill, this suggests that pup mortality was higher than normal in oiled areas in 1989.

The number of dead animals found and reported greatly underestimates the number that died. Dead harbor seals do not float like sea otters and birds, but rather sink to the bottom. Also, it is very likely that we did not learn of all the dead seals that were found. Therefore, the number of seals that died as a result of the *Exxon Valdez* oil spill was estimated based on aerial surveys conducted during the molt. Prior to the *Exxon Valdez* spill, the number of seals in Prince William Sound had been declining, with the rate of decline similar at oiled and unoiled sites (11-14% per year).

However, counts made in September 1989 indicated a significantly greater decline at oiled sites (44% compared to 8% at unoiled sites). We assumed that the trend in numbers at unoiled sites was "normal" and that any greater decline at the oiled sites was due to the spill.

Calculations indicate that there were 152 fewer seals at the oiled trend count sites than would have been expected if

the spill had not occurred. Applying the same mortality rate to other oiled sites we studied in Prince William Sound results in an estimate of 193 additional dead seals. The total of 345 seals provides some indication of the number that were killed by the *Exxon Valdez* oil spill, but it is very conservative since it includes only parts of the Sound, and it is based on only the number of hauled out seals and not the total population.

It is difficult to assess whether or when harbor seals will recover from the effects of the *Exxon Valdez* spill. We do not know why their numbers were dropping before the spill and therefore cannot predict how the additional mortality may affect the ongoing decline. Molting counts in 1992 were 34% lower at oiled sites than they were in 1988 before the spill, while counts at unoiled sites were only 18% lower.

The current trend in seal numbers in Prince William Sound as a whole is somewhat unclear. Compared with 1989, trend count data for 1992 show a 29% decline for counts made during pupping but only a 3% decline for molting counts. At Tugidak Island, counts made in 1992 were 44% lower than in 1988.

We conclude that: 1) the number of harbor seals in the area affected by the *Exxon Valdez* oil spill began declining before the spill and the population is now substantially reduced; 2) seal numbers are continuing to decline in this area; and 3) damage caused by the *Exxon Valdez* oil spill exacerbated the decline in seal numbers, at least in oiled parts of Prince William Sound.

Vital Rates and Pod Structure of Resident Killer Whales Following the Exxon Valdez Oil Spill

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The photoidentification method for assessing killer whale populations was developed by Dr. Michael Bigg, Graeme Ellis, and Ian MacAskie in the early 1970's in British Columbia. The technique was used in Prince William Sound in 1984 to develop population estimates prior to possible live capture of killer whales by Sea World Inc. Later it was used to assess the impact of longline fisheries on killer whales. At the time of the spill, detailed data on the composition of the killer whale population existed for many killer whale pods in the Sound.

In British Columbia and in Alaska two populations of killer whales exist sympatrically. The two populations, termed "resident" and "transient", have never been observed in association, nor have individuals from one population emigrated to the other. The populations also have dissimilar vocal repertoires and behaviors. Resident killer whales appear to eat primarily fish while transient whales are primarily predators of marine mammals. Social structure of transient killer whales is not well understood. Resident killer whale social structure is well described; maternal groups, the components of pods, remain together for life. This paper primarily addresses changes in the most regularly observed resident pods following the *Exxon Valdez* oil spill.

Identification photographs were taken from small vessels (less than 9 m) powered by outboard or inboard/out-

board motors which provided speed and maneuverability. Up to three vessels operated synchronously in different areas. A crew of two scientists operated each vessel, an operator/observer and a photographer.

Whales were located visually during regular searches of the study area, or by listening with a directional hydrophone, or by responding to VHF radio calls from other vessels or aircraft. Regular radio calls were made on hailing Channel 16 VHF to elicit additional whale sighting reports.

Researchers attempted to maximize the number of contacts with each pod. Although searches were made in other regions of the Sound, the greatest effort was recorded in the western Sound. This was due to the historically higher rate of encounter and larger number of sighting reports from that region.

Identification photographs were taken of the port side of the whale showing details of the dorsal fin and white saddle patch. The shape of the dorsal fin and saddle patch and observable marks and nicks are used to distinguish each whale. Photographs were taken at no less than 1/1000 of a second using high speed black and white film, Ilford HP5, exposed at 1600 ASA. A Nikon 8008 autofocus camera with 300 mm, F:4.5 autofocus lens and internal motor drive was used.

Every negative was examined under a Wild M5 stereomicroscope at 9.6 power.

All identifiable individuals were recorded. The alphanumeric code was used to label each individual. The first character in the code is "A" to designate Alaska, followed by a letter (A-Z) indicating the individual's pod. Individuals within the pod received an arbitrary number. For example, AB3 is the third whale designated in AB pod. New calves were identified and designated in their pod with the next available number. When identifications were not certain, the whale's designation was followed by a question mark and not included in the analysis. Any unusual wounds or other injuries were noted.

Recruitment of new calves generally occurs between October and April. Most new calves are first observed when field work begins in the spring or summer. Observation of calves in summer suggested recruitment to about 6 months of age. These observations were used in calculations of recruitment rates. Calves tend to stay very close to their mothers in the first year of life which permits identification of mothers with new calves. Although the white saddle patch generally does not develop for several years, other scars and marks including the shape of the white eye patch are used to reliably identify calves.

If a whale is not photographed swimming alongside other members of its maternal group during repeated encounters it is considered missing. If it is still missing during the following season it is confirmed as dead. These data were used to calculate mortality rates.

No resident whale consistently missing during repeated encounters over the course of a season has ever returned to its pod or appeared in another pod in all the years of research in Canada and the United States. In a few instances missing

whales have been found dead on beaches, but strandings of killer whales are infrequent events and most missing whales are never found. In the years from 1975 to 1987 only six killer whales were found on beaches throughout the entire Gulf of Alaska. In early Soviet research when killer whales were taken as specimens they were found to sink when shot.

However, for transient whales immigration and emigration may occur between groups. Transients missing from their original groups for periods ranging from several months to several years have been resighted swimming with other groups of transient whales. For this reason, whales missing from a particular transient group cannot necessarily be assumed dead and are described only as missing in this paper.

Births and deaths were recorded and mortality and recruitment rates calculated for each of the killer whale pods that have been repeatedly photo-documented since 1984. The AB pod, a group of 35 whales in 1988, was the most frequently encountered resident pod prior to 1989.

When this pod was photographed 6 days after the spill in April 1989, seven whales were determined missing. An additional six were missing when the pod was photographed after the winter of 1989/1990. None of these individuals have been photographed since that time and are presumed dead. One additional whale was missing in 1991 and there were no additional whales missing in 1992.

The mortality rates for AB pod ranged from 3.1% in 1988 to 19.4% in 1989 and 20.7% in 1990 followed by a decline to 4.3% in 1991 and 0 in 1992. The next highest annual mortality rate during this period was 7.7% for AE pod (13 whales

in 1988), and reflects a single mortality in 1988/89. Annual pod mortality rates on the order of 20% are unprecedented in Prince William Sound or other regions where data are available.

No new calves were recruited into AB pod in 1989 or 1990 after a recruitment of five calves in 1988. A pause in recruitment might be expected after the spike in rates that occurred in 1987/88. There is precedent in AJ pod (24 whales in 1985) of 2 consecutive years without recruitment (1985 and 1986). However, the lack of recruitment coupled with the aforementioned mortalities, reduced the number of whales in AB pod to 23 in 1990. There was one calf recruited to AB pod in 1991 and two recruited in 1992. The pod had 25 members in late 1992.

The recruitment of one calf in 1991 and two calves in 1992 and sharp reduction in the mortality rate within AB pod suggest significant change from the conditions that precipitated the mortalities in 1989 and 1990. However, the potential for additional mortalities within the pod exists. The mothers of two juvenile whales, AB38 (5 years of age) and AB41 (4 years of age) have disappeared and the longterm viability of their offspring is uncertain.

There were only two mortalities in repeatedly sighted resident pods (101 whales) other than AB pod from 1989 and 1990. These whales, AE12 and AN2, were suspected to be old, possibly post-reproductive females. They had not produced a calf since first identified in 1984. Only one of the 14 whales missing from AB pod fit this category (AB21). Of the other missing whales 8 were juveniles, 2 were maturing males and 3 were reproductive females. From 1989 through 1991 mortality rates for AB pod reproductive females was 11.1% and for juveniles it was

21.4%. Mortalities from these age classes are unusual. This is supported by data from British Columbia where the average annual mortality rate for reproductive females was 0.48% and for juveniles was 1.8% in 14 years of systematic study.

Many of the whales in the transient AT1 group (22 whales in 1989) have not been photographed since 1989. These include AT5, AT7, and AT8, the whales photographed with AT6 near the *Exxon Valdez* shortly after the spill, and AT15, AT16, AT19, AT20, AT21, and AT22. Because the pod structure of transients is not well understood and immigration and emigration between groups may occur, we cannot be certain that missing whales represent mortalities. Of note is the disappearance of AT5, AT7, and AT8 who have frequently associated with AT6 in the past. Whale AT6 was repeatedly photographed again in 1990, 1991, and 1992 without these whales. Whale AT19 was found dead on the beach in spring 1990. The status of the other non-photographed AT1 pod whales is uncertain. Also of concern is the lack of recruitment into this group since 1984.

Three killer whales carcasses were found in 1990 and one was found in 1992 on beaches in Prince William Sound. Only one was identified, AT19 in 1990. Three had marine mammal parts in their stomachs (including AT19) which suggests they were transient whales, and one had a halibut hook in its stomach which suggests a resident whale that interacted with the longline fishery or consumed a halibut with a hook in its mouth. A fifth killer whale was found on Kayak Island about 60 miles southeast of the Sound with marine mammal parts in its stomach. None of the whales had apparent bullet wounds and cause of death was not determined. There have been no

other dead killer whales observed or reported within the Sound since systematic killer whale photo-identification work began in 1983.

To summarize, the 13 whales missing from AB pod in 1989 and 1990 were confirmed as mortalities in 1991 and again in 1992. No other resident pods in Prince William Sound have demonstrated mortality rates as high as AB pod in 1989 and 1990. Most mortalities in AB pod were from age and sex classes that have demonstrated very low mortality rates in other areas. The mortality rate for AB pod has declined to zero and two calves were recruited in 1992. Recruitment rates for all other Prince William Sound resident pods have exceeded mortality rates in most recent years. Except for AB pod, all pods have maintained or increased their numbers since 1988. There has been no immigration or emigration between resident pods during that period.

There is concern that some members of the AT1 transient group are also missing or dead. However, because the social structure of transient groups is poorly understood, this cannot be established with certainty. Nine of these whales have been missing for 3 years.

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Assessment of Injuries to Prince William Sound Killer Whales

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Killer whales, *Orcinus orca*, occur in all oceans of the world. Population estimates, based on photo-identification studies, are available for the inland waters of Washington State, British Columbia, southeast Alaska, and Prince William Sound. Each area contains approximately 100 to 250 whales.

Two forms of killer whales co-exist in Prince William Sound which are known as residents and transients (Heise et al., 1992). The resident type is characterized by forming matrilineal groups within the pod. Resident pods have consistent membership overtime and have low birth and death rates (Olesiuk et al., 1990).

Birth rates are based upon the observation of new calves within a pod. Mortality rates, however, are based on the lack of observation of a known individual within a pod. The social structure of resident killer whales is such that an animal not observed for more than one year is considered dead. The social organization of transient killer whales is not well understood and the dynamic nature of these pods makes determination of their rates of mortality difficult.

The reported loss of 14 individual whales from the resident AB pod (which numbered 36 whales in 1988) for the years 1989 through 1991 (Matkin et al., this symposium) is unprecedented. Several possible explanations for the missing whales were examined.

The missing 14 animals could have been an artifact of the survey protocol. This problem was evaluated by looking

at the potential for error in the photo-identification process and the bias in survey coverage. The number of animals present in Prince William Sound pods during summer surveys in 1989-91 was obtained through detailed examination of the photographic database of individual animals. Presence or absence of members of each pod were evaluated by comparing photographs taken during the 3-year study period to previous years.

Results of the comparisons verified the absence of 14 whales in AB pod. To evaluate whether or not a mistake was made during the identification process (for example, was a whale present but mis-identified) four independent tests were conducted. Animals were recorded as being present or absent each time the pod was encountered. The results showed that earlier identifications were correct and that 14 whales were in fact missing.

Another possible bias that could have resulted in the 14 whales not being seen and photographed was the amount of effort put forth to establish presence or absence of individuals in the pod. The overall effort (miles surveyed) conducted during 1989-1991 resulted in the greatest amount of effort to date in Prince William Sound. The number of times each pod was seen in 1989, 1990, and 1991 seasons exceeded that reported for earlier studies. The amount of effort and the number of times each pod was encountered was more than adequate for locating and identifying the presence of indi-

vidual animals.

We next considered the possibility that individual whales may have moved out of the Prince William Sound area and were not available to be photographed during these studies. Although considerable searching effort took place in southeast Alaska, the missing whales were not encountered. Unfortunately, minimal effort was expended near Kodiak Island and the waters adjacent to Prince William Sound to locate the missing whales during the 1989, 1990, and 1991 seasons.

However, in 1992 researchers from the National Marine Mammal Laboratory conducted killer whale photo-identification studies from Kodiak Island to Seward, Alaska. The AB pod was not seen during these investigations. An examination of the 20-year killer whale database from British Columbia and Puget Sound, Washington, indicated that no resident killer whale consistently missing during repeated encounters had ever returned to its pod or appeared in another pod.

The possibility that the missing whales have moved out of the area is not supported by our knowledge of the social structure and behavior of resident killer whales. Based upon the historical life history information, it is likely that the missing resident whales are dead and have not moved off to other areas.

However, a perturbation as severe as the *Exxon Valdez* oil spill and its direct impact on cetaceans has never before been investigated. It is therefore possible that a major catastrophe such as the *Exxon Valdez* oil spill could have affected killer whales in ways never described before. This possibility, although highly unlikely, should not be disregarded.

The most reasonable explanation for

the disappearance of the 14 whales is that they are dead. However, the cause(s) of their death remain unclear. Natural mortality is certainly plausible, but unlikely. This species is characterized by a low birth and death rate (less than 2.2% per year or less; Olesiuk et al., 1990). The mortality rate for AB pod calculated for the 1989 season with the loss of seven whales was 19.4%. Six additional whales were reported missing from AB pod resulting in a 20.7% mortality rate for the 1989/90 season. In 1991, one more whale was noted as missing from AB pod (mortality rate of 4.3%).

These rates for the 1989 and 1990 season are significantly higher than would be expected from natural causes. It is unlikely that natural mortality would account for more than 1-3 animals, and not the loss of 14 whales over a 3-year period as observed.

Examination of other causes to explain the mortality of the 14 missing whales are complicated by the past history of AB pod. This pod was involved in interactions with the Prince William Sound sablefish longline fishery in the mid 1980's. In 1985, we received reports of killer whales being shot at by fishermen. Several of the animals showed evidence of bullet wounds. In 1985, three whales were reported missing. In 1986, three additional whales were gone. In 1987 and 1988, this pod lost two more individuals. The loss of at least some of these 8 whales was attributed to shooting (although never confirmed). These whales were not seen again after the year they were first identified as missing.

It is possible that the 14 whales reported missing during the 1989 through 1991 season could have been shot. However, this is unlikely because (1) longline fishing was closed between the time when

all whales were accounted for (September 1988) and the time when the first seven whales were first determined missing (March 1989), (2) there were no reports of shootings and, (3) no new bullet wounds have been observed on individuals of AB pod since 1986.

The remaining cause of death considered was the effect of the oil spill. Six different killer whale pods were observed transiting directly through oil (light sheen) but only AB pod suffered losses. The loss of the first seven animals from AB pod could have been through direct contact with the oil, such as from inhalation of toxic volatile gases or ingestion. The loss of the six additional whales one year later is more difficult to explain from oil effects, but might have been associated with residual effects or from indirect effects (e.g., eating contaminated prey).

None of the missing whales were found stranded, although killer whales typically sink upon death. Four carcasses (only one whale could be identified and it was not from AB pod) were found during the three-year period (1989-1991). This stranding rate is high compared to other geographical areas, and from previous stranding rates from the Prince William Sound region. However, this may simply have been an artifact of increased effort after the spill. Blubber samples and scrapings from the stomach lining from the stranded whales were analyzed for hydrocarbons. There was no indication of oil contamination in these tissues and cause of death could not be determined. Caution, however, must be used when interpreting these results since the carcasses were old when found and

decomposition decreases the viability of the tissue samples for hydrocarbon analysis.

In conclusion, the cause(s) of the deaths of 14 killer whales from AB pod is unknown. We are confident that (1) whales have not been mis-identified, (2) adequate effort was made in Prince William to locate the missing animals, and (3) the number of encounters was sufficient to evaluate the presence or absence of an individual whale. The current life history information available on killer whales precludes the possibility that the whales moved elsewhere.

Therefore, we assume that the whales are dead from either, or a combination of, natural causes; a result of interactions with fisheries; or, for the *Exxon Valdez* oil spill. The highest mortality rate ever reported in the literature for North Pacific resident killer whales occurred in 1989 and 1990, coinciding with the *Exxon Valdez* oil spill. There is a strong correlation between the loss of the 14 whales and the *Exxon Valdez* oil spill, but there is no clear cause and effect relationship.

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Humpback Whale Abundance and Distribution in Prince William Sound

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Humpback whales, *Megaptera novaeangliae*, number about 10,000 animals worldwide, of which perhaps 1,500 occur in the North Pacific (Baker and Herman, 1987). They are currently listed as an endangered species under the U. S. Endangered Species Act of 1973. During winter the North Pacific population occurs principally off Hawaii and Mexico. During summer they range widely across the North Pacific with concentrations off Kodiak Island, in Prince William Sound and in southeast Alaska. Prince William Sound is considered a major feeding area for the North Pacific stock of humpback whales.

Identification of individual whales is possible through use of ventral fluke coloration patterns and natural marks. Researchers conducting photo-identification studies in Prince William Sound and southeast Alaska had collected a substantial photographic database on humpback whales prior to the spill.

Based on this historical database, at least 50 individual animals were known to occur annually in Prince William Sound and approximately 300 whales from southeast Alaska. Not all identified whales are seen each year in each area, however, site fidelity has been reported. Whales seen in Prince William Sound are generally not seen elsewhere, and whales seen in southeast Alaska are not seen in Prince William Sound. Over the years, however, three whales have been seen in both Prince William Sound and south-

east Alaska.

The objectives of this study were to (1) count and individually identify humpback whales in Prince William Sound and southeast Alaska; (2) test the hypothesis that humpback whale distribution and abundance within Prince William Sound and adjacent waters is similar to that reported for years prior to 1989; and (3) test the hypothesis that humpback whale natality and mortality in Prince William Sound has not changed since the oil spill.

During 1989 and 1990, photographs of individual humpback whales occurring in Prince William Sound were collected from May to September each year to address objectives one and three. To determine if significant changes occurred in whale distribution (objective two), concurrent photo-identification work was carried out in 1989 in southeast Alaska.

Photographic analysis of Prince William Sound humpbacks documented 59 individual whales during the 1989 season. In southeast Alaska, 552 individual whales were identified. During 1990, 66 individual whales were documented in Prince William Sound. The 1989 and 1990 counts represent the largest number of humpback whales ever photographed in Prince William Sound. The effect of increased observer effort during these two seasons, however, must be considered. In particular, the high levels of effort during these two years have clearly increased the number of indi-

vidual humpback whales identified relative to numbers prior to 1989.

The distribution of whales in Prince William Sound during the 1989 season was compared to the known distribution prior to the oil spill. Distributional data prior to 1989 noted concentrations of whales in the Lower Knight Island Passage area. In 1989, fewer whales were observed in this area. The reasons for this are unclear, but may be related to increased vessel and aircraft traffic in 1989, or may simply be caused from natural fluctuations in prey. In 1990, whales were again abundant in the Lower Knight Island Passage area, similar to their distribution prior to 1989. Despite considerable effort in 1989, humpback whales known to occur in Prince William Sound were not observed in southeast Alaska.

No apparent shift in distribution within Prince William Sound was noted in 1990; whales were again abundant in the Lower Knight Island passage area, similar to their distribution in 1988. De-

spite considerable effort in 1989, humpback whales known to occur in Prince William Sound were not observed in southeast Alaska.

The combined average annual reproductive rate for 1980 through 1988 for Prince William Sound humpback whales was 9.4%. In 1989, the reproductive rate was 6.3%; in 1990 it was 10.8%. Seven out of the eight females present with calves in 1990 had been photographed in Prince William Sound in 1989, and thus these whales were pregnant at the time of the spill. No reports of dead stranded humpback whales occurred within Alaskan waters during this two-year period.

From the available data, it does not appear that the *Exxon Valdez* oil spill had any measureable impact on the North Pacific humpback whale population.

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Occupational Exposures from Oil Mist During the Exxon Valdez Spill Cleanup

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Crude oil cleanup during the *Exxon Valdez* spill relied heavily on high pressure water and steam (Exxon, 1989) which generated an oil mist. Monitoring records document an average oil mist exposure 12 times in excess of permissible exposure limits.

The National Institute for Occupational Safety and Health (NIOSH) reported 1,811 worker's compensation claims in 1989 related to the *Exxon Valdez* oil spill (Gorman et al., 1991). The leading non-physical injury reported was respiratory system damage. Inhalation of oil mist is well recognized as a cause of occupational respiratory damage (Lancet, 1990; Robertson et al., 1988).

Prior evaluations of the 15,000 *Exxon Valdez* cleanup workers occupational exposure to airborne contaminants stated that, in general, exposures were a fraction of permissible exposure limits (PEL), (Gorman et al., 1991; Wade, 1990). However, oil mist measurements were not mentioned by Wade and NIOSH conducted limited oil mist testing. This is the first independent review of *Exxon Valdez* oil spill cleanup worker exposure records.

The objective of this study is an evaluation of the health and safety implications of using hot water and steam at high pressure and elevated temperature to clean crude-oil-contaminated beaches.

Under contract to Exxon, Med-Tox Associates collected over 6,000 air samples in 1989 from *Exxon Valdez* oil spill cleanup workers. The Alaska Health

Project obtained Exxon and Med-Tox exposure data, health and safety records, and laboratory procedure manuals. This information was then compared to the literature.

The data collected from workers revealed that the average exposure exceeded the NIOSH limit by 12-fold. The maximum overexposure of 400 times the PEL was found on a "hot wash beach." Average exposures for other chemicals were below NIOSH recommended PEL. However, maximum exposures were significantly greater than NIOSH limits; that is, total PNAs 170 times greater than the limit, benzene 160 times greater, hydrogen sulfide 40 times greater, butoxy-ethanol eight times greater, and carbon monoxide six times greater. NIOSH limits were adjusted for the increased length of working day, as recommended by Exxon publications (see "Extended Work Days" below), but were not adhered to.

Another issue of particular concern is the fact PEL are developed on a chemical-by-chemical basis and Exxon did not take into account multiple simultaneous exposures with synergistic potential. Finally, the upper 5% of exposures, in every case listed below, exceeded NIOSH limits.

Three serious problems are evident with Exxon's laboratory procedures and data interpretation regarding oil mist monitoring records: standard reference material, applicability of PEL, and extended work days.

Standard Reference Material

The standard reference material for oil mist PEL is "mineral oil" (NIOSH, 1990). Mineral oil is a highly purified product designed for non toxicity and freedom of irritation to humans and use in the preparation of pharmaceuticals (ASTM, 1989). Oil spill cleanup workers were exposed to Prudhoe Bay crude oil (PBCO). PBCO consists mostly of aliphatic and aromatic components and smaller amounts of heterocycles and asphaltenes. The aliphatic fraction is dominated by n-alkanes containing 11 to 40 carbon atoms and isoprenoid hydrocarbons. The aromatic components consist of a series of parent and alkylated naphthalenes, phenanthrenes, fluorenes, biphenyls, chrysenes, and benzantracenes (Rahimtula, 1987).

In addition to hydrocarbons crude oil contains sulfur compounds such as thiols, sulfides, disulfides, and thiophenes. The higher boiling point sulfur compounds are thiocyclo-, thiobicyclo-, thiotricycloalkanes, complex thiophenes, and benzothiophenes. Basic nitrogen compounds found are pyridines and quinolines while nonbasic nitrogen compounds include pyrroles, indoles, and carbazoles. Oxygen compounds found include ketones and phenols with alkane and cycloalkane acids in the higher boiling point fraction (Costantinides and Arich, 1967).

Nickel and vanadium occur primarily as complexes such as porphyrins and over 30 metals commonly occur in crude oil. Other substances are introduced into crude oil during the process of drilling, pumping, preparing and transportation (IARC, 1989). Although mineral oil may be derived from crude oil, the refining process selectively removes a specific hydrocarbon fraction

leaving most of the components mentioned above in other residues.

Based on the differences between mineral oil and PBCO two things should have been done regarding calibration standards and PEL. Laboratory equipment should have been calibrated using PBCO as the standard. A review of laboratory procedures and quality control documentation did not find evidence that PBCO was used as the standard for which to measure PBCO derived oil mist (Pristas, 1989). Substantial bias likely exists in the data because of inappropriate use of standard reference materials. The problem of inappropriate standards and infrared spectrophotometric quantification of crude oil is well documented (Baugh and Lovegreen, 1989).

Applicability of PEL

No corrections were applied to PEL for the elevated toxicity of crude oil compared to mineral oil. Crude oil is a carcinogen, neoplastigen and tumorigen when applied to the skin. Inhalation of vapor or particulates can cause aspiration pneumonia (Sax, 1989). A material safety data sheet for crude oil recommends a PEL of 0.2 mg/m³ (Lyondell, 1990); 25 times lower than the 5.0 mg/m³ PEL selected as relevant by Exxon. NIOSH recommends a PEL of 0.1 mg/m³ (NIOSH, 1990); 50 times lower than Exxon's.

Extended Work Days

Finally, the working conditions during the *Exxon Valdez* cleanup were not 8 hour days with weekends off. More typically, workers were on the job in excess of 12 hours a day, seven days a week and some for months without a break. Exxon recognized more than ten years ago that the PEL for airborne toxicants were probably inappropriate with-

out modification for unusual work shifts. A simple linear equation was proposed by Exxon as a first step toward health and safety concerns (Exxon, 1986). However these considerations were not taken into account for the extremely long shifts of *Exxon Valdez* spill cleanup workers. If we apply Exxon's model to NIOSH oil mist PEL, the acceptable limit should be reduced by a factor of at least 2.1 (84 vs. 40 hour week). Using the PEL of 0.1 mg/m³ and a factor of 2.1 yields a PEL of 0.05 mg/m³.

The average worker was exposed to 12 times more oil mist than what NIOSH standards permit. Some exposures were 400 times higher than PEL. Whether or not an individual worker's health problem was caused by over exposures during the *Exxon Valdez* cleanup can only be determined on a case-by-case basis. Based on the information summarized above, further research is needed regarding medical histories of exposed workers to protect future generations when selecting cleanup technologies at other spills.

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Boat-Based Surveys of Sea Otters (*Enhydra lutris*) in Prince William Sound, Alaska.

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When the T/V *Exxon Valdez* ran aground on Bligh Reef on March 24, 1989, the resulting spill of 11 million gallons of crude oil into Prince William Sound resulted in the death and injury of more than a thousand of sea otters (*Enhydra lutris*). As part of the Natural Resources Damage Assessment effort, the U.S. Fish and Wildlife Service conducted boat-based population surveys of marine birds and sea otters in Prince William Sound between June, 1989 and July 1991. Based in part on similar surveys conducted during the summers of 1984 and 1985, the purpose of this work was to estimate post-spill sea otter abundance in order to determine initial injury to the population, and monitor continuing injury or recovery.

The study area consisted primarily of the waters of Prince William Sound, Alaska. The study area was divided into three distinct strata: shoreline, coastal, and pelagic. The shoreline stratum was based on shoreline transects surveyed by Irons, Nysewander and Trapp (1988) during the summers of 1984 and 1985, and was defined as the 200 m-wide strip immediately adjacent to the coastline. Within the Prince William Sound study area, 742 shoreline transects were defined with a total area of 822.3 km².

Waters outside the shoreline stratum were divided into sampling "blocks" based on a 5-minute latitude/longitude grid system. These blocks were then stratified into two categories: coastal and pelagic. The coastal stratum was com-

prised of those blocks that are immediately adjacent to 1 km or more of shoreline, while the pelagic stratum was comprised of those blocks that are adjacent to less than 1 km of shoreline. This classification scheme resulted in the creation of 207 coastal and 86 pelagic blocks, with total areas of 4,524 km² and 3,637 km², respectively. Within each block, a number of 200 m-wide strip transects (usually two) were systematically placed and sampled.

Watercraft used in this survey were 25' Boston Whalers, with three crew members serving equally as operator and observers. Shoreline transects were surveyed from 100 m offshore at a cruising speed of 5-10 knots. One observer scanned the water from the vessel up to and including the shoreline, while another observer scanned the water from the vessel seaward an additional 100 m. Coastal and pelagic transects were surveyed at a slightly faster cruising speed of 10-15 knots, with each observer scanning the water from the trackline of the boat outward 100 m. In addition, the watercraft operator assisted with observations of animals directly ahead of the vessel. While the vessel was in motion, all marine mammals and birds sighted were recorded on standardized data sheets.

As stated earlier, the shoreline stratum was based on a set of transects originally surveyed during the summers of 1984 and 1985 (Irons et al. 1988). Over the course of two field seasons, virtually all

of the available shoreline habitats were surveyed (708 out of the possible 742 transects). These data served as the baseline for comparison with post-spill surveys.

Post-spill surveys were conducted in June, July, and August of 1989, March, June, July, and August of 1990, and March and July of 1991. Approximately three weeks were needed to complete each replicate of the survey. Post-spill surveys were initially conducted during the summer of 1989 as a random sample of approximately 25% of available shoreline transects and the original coastal and pelagic blocks. Only the shoreline stratum was sampled during June 1989. All three strata were sampled in each of the remaining surveys. Once the initial random sample of transects and blocks was chosen, each successive survey replicated the same sampling units.

Classification of sampling units as oiled or unoiled was based on Alaska Department of Environmental Conservation overflight data collected at the time of the spill (ADEC 1989). Aerial observations were used to create a GIS coverage depicting the movement of oil over the surface of the water. Since sea otters are highly mobile animals, otters inhabiting areas adjacent to the path of the oil could have encountered the slick during their normal movement patterns. Given this fact, coupled with an inherent uncertainty as to the exact geographical extent of the surface oiling, a buffer zone of 5 km was added to the ADEC overflight data coverage to represent an area within which otters might have been affected by oil. Shoreline transects, and coastal and pelagic and offshore blocks with any area located within 5 km of surface oil were classified as oiled.

Sea otter density and abundance esti-

mates for each survey strata were calculated using ratio estimator techniques (Cochran 1977).

In the unoiled area, otter densities in the shoreline stratum increased 13.5% between the pre-spill surveys of Irons et al. (1988) and the summer 1989 surveys. Otter densities in the oiled shoreline stratum declined approximately 34.6% during the same period. Surveys conducted in the summer of 1990 show further declines in shoreline sea otter density within the oiled area. However, otter density in unoiled areas also exhibited a decline during the same period. Otter density within the oiled area did not appear to have changed between July 1990 and July 1991. With the exception of the July 1990 survey, otter densities in the oiled area were consistently lower than those in the unoiled area, which is in contrast to the pre-spill pattern.

In the unoiled area, there was considerable overlap between pre- and post-spill shoreline population estimates. There was no overlap between pre- and post-spill estimates in the oiled area for this stratum. This lack of overlap between estimates within the oiled area suggests that the post-spill population was significantly lower than the pre-spill level.

Although sea otter densities were lower in coastal and pelagic strata, given their larger total areas, these strata contained a considerable number of otters. In some instances, these strata accounted for over 50% of the total estimated population. Post-spill population estimates from this study range from a high of 8,242 ($\pm 2,280$) in July 1989 to a low of 4,399 (± 948) in March 1991.

In July 1990, it was the decision of the Management Team that data from the various damage assessment studies

should be brought together in an attempt to quantify initial injury to the Prince William Sound sea otter population. In a cooperative effort, results of studies on sighting probability, carcass recovery rates, and the age structure of the recovered carcasses were synthesized with these survey data to calculate an estimate of the initial first-year injury (Garrott, Eberhardt, and Burn 1992). This exercise produced a loss estimate of approximately 2,800 sea otters for Prince William Sound (Garrott et al. 1992). Population trends in the oiled area suggest that additional losses may have occurred beyond the first year, but have not been quantified at this time.

Although estimates of shoreline sea otter density within the oiled area fell well below their pre-spill values, it is important to note that a substantial fraction of the population survived the spill and its aftermath. One reason for this may have been the presence of small bays and coves that remained relatively oil-free (Garrott et al. 1992). In the southwest portion of the Sound for example, Bainbridge Passage was heavily oiled. Yet during each of the three surveys conducted in June, July, and August 1989, we observed 30-40 otters concentrated in an apparently unoiled cove on the southern side of the Passage. These unoiled refuges were scattered throughout the spill zone, and may have provided a haven for otters.

The long-term effects of the spill on sea otters in the western portion of Prince William Sound are unknown. Two key factors that will determine those long-term effects on sea otters will be the impact of the spill on the populations of sea otter prey items (primarily mussels and clams), and continued exposure of sea otters to hydrocarbons through their

prey. Either one or both of these factors could have an impact on the recovery of the sea otter population within the oiled area of the Sound.

As a means of estimating the Prince William Sound sea otter population, this survey suffered from a lack of precision. Most of this variability in the estimates came from the coastal and pelagic strata. Although 25% of the blocks within these strata were sampled, only 10% of the area within the blocks themselves were surveyed. The net result of this design was that every otter sighted in the coastal and pelagic strata equated to roughly 50 otters in the final estimate.

In order to monitor population trends with respect to recovery, I believe that the shoreline stratum is the best means of judging the status of the population. The majority of otters sighted during this survey were observed on shoreline transects, making this stratum a good index of population size. It is also the only area for which pre-spill data exist. One potential criterion for estimating when the sea otter population is fully recovered, is that point when sea otter densities within the shoreline stratum increase to pre-spill levels.

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Hydrocarbons in Mussels and Subtidal Sediments: Graphical Presentation of Hydrocarbon Analysis Data With Geographic Map Data

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Oil movements on the surface were tracked during the spill, but there were large unknowns as to what the subsurface fauna were exposed to. Several studies sampled mussels and subtidal sediments for hydrocarbon analysis in an attempt to characterize the oil exposure at sites and habitats appropriate to individual projects. This project attempts to map oil exposures geographically and temporally, using samples collected by many projects.

Maps displaying the distribution and persistence of *Exxon Valdez* crude oil through presentation of sediment and of

mussel tissue hydrocarbon analysis data on shoreline oiling maps will be presented, together with an interpretation of the hydrocarbon data symbols used.

A map for each project that had sediments or mussel tissues analyzed will be available for public inspection, with at least one map for each project containing all the hydrocarbon data generated for each year of the project. This poster session will focus on the data manipulation and quality assurance steps from the raw data generated by the chemistry laboratories through the GIS mapping software.



Management of Natural Resource Damage Assessment Samples and Analytical Data

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Within 48 hours after the grounding of the *Exxon Valdez*, samples were being collected by State of Alaska and Federal Trustee Agencies (Alaska Departments of Fish and Game, Environmental Conservation and Natural Resources; U.S. Department of Agriculture, Forest Service; U. S. Department of Commerce, National Oceanic and Atmospheric Administration; and U. S. Department of Interior, Fish and Wildlife Service) to document the exposure of natural resources to the spilled oil and to provide a basis for the determination of the effects of the oil. During the course of the Natural Resource Damage Assessment, over 36,000 samples of water, biota and sediment were collected from Prince William Sound and the Gulf of Alaska to meet these objectives. An additional 1,500 experimental samples were generated in various laboratory experiments.

A cooperative project between the U.S. Fish and Wildlife Services (F&WS) and the National Oceanic and Atmospheric Administration (NOAA), Natural Resources Damage Assessment (NRDA) Project Technical Services #1, was responsible for (1) archiving and tracking of these samples; (2) analysis of the samples, including selection of samples for analysis, and the development and implementation of an analytical quality assurance plan which defined criteria for the quality and acceptability of the data; and (3) management of the analytical data. A relational database (PWSOIL) was used to carry out these

tasks, i.e. to maintain and manipulate all data and information related to the collection and analysis of samples for petroleum hydrocarbons.

PWSOIL is based on the design used by NOAA's National Status and Trends Program, modified for similarity to other databases either already in use or being planned by F&WS and the U.S. Environmental Protection Agency at the time of the grounding of the *Exxon Valdez*. The original design was then modified in an iterative fashion, however, to meet newly defined needs and objectives. All changes to the design of PWSOIL and to the data and information maintained by this database were preceded - and often initiated - by discussion with the users, the Project Leaders. PWSOIL is supported by hard copies of all chain-of-custody records and all developed analytical data, which themselves are kept under chain-of-custody procedures. A complete description of the design, including definitions of the variables and instructions for users, and implementation of PWSOIL is documented in Manen et al.

PWSOIL is focused on the unique identification number assigned to each sample by the database manager. Associated with this number are the data and information which describe and identify the sample, e.g. where the sample was collected (place name and latitude/longitude); who collected it; when it was collected; how it was collected; the sample identification number assigned by the collector; what kind of sample it is (sedi-

ment, water or tissue); if a tissue sample, what kind of tissue and from what plant or animal; if a sediment or water sample, the depth at which it was collected; and whether or not it has been analyzed. All of these variables are provided by the collector or Project Leader as part of the chain-of-custody record. The maintenance of these variables in PWSOIL allows the sorting of samples by location, project, species, etc.

Over 12,000 of the NRDA samples have been analyzed. The majority of the samples were analyzed for petroleum hydrocarbons (73 parameters) by GC/MS at Texas A&M University; smaller, specialized sample sets were analyzed at NOAA's Auke Bay Laboratory (ABL) and NOAA's Northwest Fisheries Center (NWFC). Semi-qualitative, non-compound specific information was developed for some sediment samples as well as the concentrations of petroleum metabolites in the bile of fish, birds, terrestrial and marine mammals by UVF, a technique to determine the presence of petroleum hydrocarbons. These analyses were performed by Texas A&M University and NOAA's NWFC. All data resulting from these analyses, as well as (1) calculated indices and parameters; pristane/phytane ratio, carbon preference index, saturated hydrocarbon weathering ratio, and sums of the alkanes, aromatics and hydrocarbons; (2) supporting data; grain size, total organic

carbon, percent moisture and surrogate recoveries and (3) quality control data; the results of the analysis of blanks, standard reference materials and in-house control materials are maintained by PWSOIL in a batch fashion associated with the sample identification numbers.

The use of PWSOIL to maintain and manipulate the analytical data in program-defined reporting formats and file structures allows the use of data across projects and analytical laboratories. PWSOIL has supported data analysis for all individual NRDA Projects which collected samples for hydrocarbon analysis; forms the basis of the secondary database developed by NRDA Project Subtidal #8, and has been used by NRDA Project Technical Services #3 to develop GIS (mapping) products and by Exxon in continuing litigation. To facilitate access to this database and these data; the database (PWSOIL), Users' Manual, and supporting analytical and quality assurance documentation have been made available as a stand-alone CD-ROM (Compact Disc-Read Only Memory). This CD uses non-proprietary software developed by U.S. Geological Survey and is the result of a cooperative project between NOAA and the Geological Survey.

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Pre-spill and Post-spill Concentrations of Hydrocarbons in Sediments and Mussels in Prince William Sound

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This study provides comparison of petroleum hydrocarbon concentrations in mussels and sediments before and after the *Exxon Valdez* oil spill. Within several days of the spill in 1989, mussels (*Mytilus trossulus*) and sediments at six historically (1977-1981) established intertidal baseline sites in Prince William Sound were resampled. Additionally, six sites were established along the spill trajectory before oiling, and sampled both before and after oiling to measure changes in petroleum hydrocarbon levels in sediments and mussels. Sampling continued in 1990 and 1991.

Both mussel and sediment transects were 30 m long and usually set parallel to the water line. The sediment transect was generally down slope from the mussel bed in finer grain material. Triplicate (30 each) samples of mussels were taken along the transect line; and triplicate samples (each a composite of 10 subsamples) were collected of sediment. All samples were placed in hydrocarbon-free jars and frozen according to established protocol.

There were no detectable polynuclear aromatic hydrocarbons (PAHs) in mussels sampled prior to the *Exxon Valdez* oil spill (1977-1980) and levels in sediments at 4 of these sites were low, generally under 20 ng/g dry weight. The pattern of low hydrocarbon levels in both mussels and sediments continued after 1989 at sites not impacted by the spill.

Sleepy Bay, a heavily oiled site, had PAH concentrations in sediments nearly

100 times historical levels (established for other sites in the Sound) in May, 1989 (939 ± 404 standard error ng/g dry weight—an amount approximating 6% *Exxon Valdez* crude oil). Sediment PAH's at this site declined to 168 ± 17 ng/g in 1990 and 42 ± 4 in 1991. Two other oiled sites (Bay of Isles and Elrington Island) showed increases (10-20 fold) of aromatic hydrocarbons in sediments in 1989, decreases in August, 1989, and increases in April, 1990. By August 1990, PAHs in sediments from Elrington Island had decreased to values similar to unoiled sites, but Bay of Isles sediments remained elevated over unimpacted sites. Differences in exposure to wave action probably accounts for these variations in recovery. Both Sleepy Bay and the Elrington Island site are quite exposed to wind and wave action while the Bay of Isles site at the southern tip of the South Arm is quite protected from inclement weather.

PAH's in sediments from most of the other sites, Bligh Island, Naked Island, Olsen Bay, Siwash Bay, and Perry Island, were detected at levels not elevated from historical concentrations.

Mussels from Sleepy Bay, the South Arm of the Bay of Isles, and the Fox Farm on Elrington Island all showed high PAH concentrations in 1989 (up to $143,000 \pm 13,900$ ng/g dry weight). These levels had decreased to $174 \pm 27.0 - 21,700 \pm 1,500$ ng/g in 1990 and $166 \pm 16.0 - 5,960 \pm 1,100$ ng/g in 1991. Mussels from Naked Island and Crab Bay in Sawmill

Bay showed elevated PAH concentrations ($1,950 \pm 135$ - $6,990 \pm 247$ ng/g) in April and May of 1989. PAH's in mussels from Olsen Bay, Bligh Island, Barnes Cove and Siwash Bay were usually detected only sporadically at concentrations near detection limits (about 10 ng/g dry weight for individual PAH's); naphthalene and substituted naphthalenes were the most frequently detected PAH's at these sites. Mussels from the latter 5 sites

had between 60.0 ± 5.00 and 243 ± 27.0 ng/g during 1990 and 1991.

Maps created by GIS systems detailing extent and concentrations of PAHs will be displayed.

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Surface Modeling of Floating Oil, The 1989 *Exxon Valdez* Oil Spill, Prince William Sound, Alaska

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Shortly after the *Exxon Valdez* ran aground on Bligh Reef in the early morning of March 24, 1989, response teams were formed to collect information on oiling, to protect the priority areas, and to begin clean up. Due to adverse weather, difficult logistics, and the size of the spill, observational information collected during this time on surface oiling was limited. Unlike shoreline oiling, surface oiling cannot be surveyed at a later time.

Through a series of modeling efforts by National Oceanic & Atmospheric Administration—Hazardous Materials Response Branch (NOAA) and the Natural Resource Damage Assessment—Technical Services 3 (TS3), surface oiling could be estimated for general purposes. NOAA has designed a trajectory hindcast model, called the On-Scene Spill Model (OSSM), which estimates the flow of oil based on wind and current patterns.

Technical Services 3 is an inter-agency group composed of geographic information systems (GIS), and technical staff

from the Alaska Department of Natural Resources and the U. S. Fish and Wildlife Service. TS3 used the OSSM results in a geographic surface model, called Triangular Irregular Network (TIN). As a result of the OSSM and TIN models, a series of maps were produced illustrating the general flow over time and the relative concentration of the oil.

This poster explains and illustrates what information and advanced techniques were used to create a modeled surface oiling map. In addition, the first 2 weeks of the spill will be represented for Prince William Sound.

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Integration of Shoreline Oiling Data Sets

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Since the *Exxon Valdez* oil spill disaster of March 1989, much attention has been focused on the beaches of southcentral Alaska. Multiple federal, state, and local agencies collected information for both the response effort and the damage assessment process. Because of the number of agencies involved and the urgency of data collection, much of this information was captured using a variety of digital mapping techniques and tools. As a result, many agencies had their own digital interpretations of the coastline.

When analysis began, many of the damage assessment and restoration teams needed to know the extent of the oiling based on a specific geographic criteria, such as, land ownership (state, federal or private) or environmental sensitivity. However, before this type of spatial analysis could be done, an integrated data set containing the selection criteria and shoreline oiling needed to be created.

Data Integration

After reviewing different software packages, our findings were that shoreline data integration was difficult at best, with any software. Initially, highly interactive attribute transfer procedures were used, but they were very time consuming and subject to operator error. This type of manual processing would not suffice for an exercise of this dimension.

We looked for an automated method for producing large-scale shoreline integration across multiple data sets with

available software tools. We found that in using available software, all the shoreline data sets would have to be coincident in their shape and contain segments of equal length. Even with the best control practices available, it would be difficult to ensure such extreme levels of accuracy across multiple data sets. Because of the large geographical extent, multi-agency participation, and the variation between the data sets, we found no automated process existed to provide multiple data set integration with a high degree of geographical accuracy. The only other option was to design a process that would allow large scale data set integration, while providing a high degree of accuracy.

Methodology

While direct, large scale line-on-line integration was not available, tools do exist for large scale polygonal integration, such as polygon-to-point, polygon-to-polygon, and polygon-to-line. Technical Services Study No. 3 developed a process to spatially convert selected linear (shoreline) data sets into polygon data sets so that currently available tools could be used for data set integration.

The specialized software that was developed used a three-fold process to convert the shoreline data set into a polygon data set. First, a raw polygon data set was created whose width was tailored to coincide with the shoreline data set and reflect the differences in spatial resolution between the various shoreline data sets that were to be integrated. Sec-

ond, adjacent polygon boundaries were created to delineate the changes in the original shoreline data set. Polygon label points were also created to match the original line data set. When the shoreline attributes are transferred to the polygon label points, the polygon data set is spatially equal to the shoreline data set from which it was made (McMahon, et al 1991).

Third, once the polygon data set was completed, attributes were integrated, or transferred, onto a common shoreline using polygonal integration algorithms (ESRI 1990). The transferred attributes spatially match the original shoreline data set.

Final Verification

Plots were used for a spatial check by visually comparing both the original shoreline data set and the new integrated data set for accuracy. A quantitative analysis between the original data set and the integrated data set showed an average difference shoreline oiling totals of less than 1%. Spatial analysis showed maximum shoreline deviation of between 2 and 3 meters in less than 3% of the shoreline. Maps of integrated data sets were also reviewed for accuracy by field investigators. At the standard scale that these maps were produced, these differences would not be seen.

This process started with the integration of two major coastal themes: shoreline oiling and environmental sensitivity. Many of the NRDA investigators needed to focus specifically on areas of high environmental sensitivity. Maps and statistical summaries were produced of specific sites where habitat protection and analysis was of concern. These results were also used to provide NRDA investigators and peer reviewers with estimates of total oiled shoreline by shoreline sensitivity.

Because some government agencies focused primarily on oiling to federal or state lands only, we combined land status into the integrated data set to provide ownership damage assessment.

One of the more interesting views of the combined data set was in the form of shoreline oiling change. By looking at the difference in shoreline oiling between various survey years, changes in shoreline oiling become apparent. Some areas indicate less oiling from year to year, possibly due to mechanical and/or natural cleaning. While other beach segments show an increased amount of shoreline oiling, possibly due to the re-floating and beaching of oil between surveys. This is a very interesting look into beach dynamics.

Data Sources

The 1989 oiling data was delivered from the oil spill response staff of the Alaska Department of Environmental Conservation (ADEC 1989; 1990a; 1990b; 1990c). The spring 1990 shoreline survey (SSAT) was digitized by Exxon from multi-agency field reports. Environmental sensitivity index maps were produced by Research Planning Institute (RPI, 1979, 1983a, 1983b, 1985, 1986) for the National Oceanic and Atmospheric Administration and digitized by Environmental Services Research Institute.

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A Reconstruction of Pink Salmon Wild Stock Runs in Prince William Sound

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Assessing the effects of the *Exxon Valdez* oil spill on the Prince William Sound wild pink salmon (*Oncorhynchus gorbuscha*) fishery requires knowledge of the spatial and temporal distributions of the individual salmon stocks. This knowledge is important because geographic and economic factors dictate that the harvest occur in mixed-stock areas. The incidence of large hatchery runs co-occurring with the wild stocks in the fishery and the lack of information on individual stock contributions to the catch in these areas make the management task more difficult. Estimates of stock-specific catches are required to determine the abundance of individual stocks. Run reconstruction, known as the "poor man's stock ID," builds a history of a stock's movement through a fishery, providing stock-specific information with few data requirements.

We develop a multi-stock/multi-district run reconstruction of wild pink salmon stocks using catch, effort, tagging and escapement data to estimate stock-specific run characteristics. The reconstruction works backward beginning with the fish in their spawning streams and projecting them backward through the fishery to the time when they enter the sound. This method is preferable to a forward projection because it requires fewer assumptions about the distribution of entry to the fishery. The completed reconstruction provides a seasonal history of each stock by estimating daily stock abundances in each

district, stock-specific contributions to the catch in each district, district-specific catchabilities, movement rates between districts and initial abundances of each stock.

A continuous database of daily catch and effort and weekly escapement records in Prince William Sound exists for the years between 1968 and the present. It is necessary to identify and remove hatchery contributions from the catch records. This can be accomplished for recent years with coded wire tag information. Effort is defined as one seine boat fishing for one day. Purse seine fishing gear is used to harvest pink salmon in the sound except in District 223 where a gillnet fishery co-occurs with the seine fishery. In this case the gillnet effort is standardized to seine effort units. Information from coded wire and radio tagging experiments is used to establish salmon movement and rates. To avoid undue complexity, a stock is defined as all the pink salmon spawning in the streams of a management district. For the purposes of damage assessment and because the data are more accurate, we reconstruct the 1990 and 1991 pink salmon runs.

We begin the reconstruction with the daily entry of fish into the streams assuming that a fish enters only one stream and remains there until it dies. In Prince William Sound, salmon hold at the stream mouth for a number of days before ascending. During this time they are not subject to harvest. Accordingly, the

stream entry for stock is delayed before being backed into the fishery as escapement.

We model the migration of stock using a transition matrix, assuming that salmon movement in Prince William Sound is directed, stock-specific and constant over the season.

The sum of the daily stock-specific catches is the daily catch in that district. Because of the backward nature of the reconstruction, catches are modeled as an inverse survival, becoming additions to the stock abundances. This conveniently avoids the possibility of removing more fish than are available. We accumulate the salmon in pools in each district as we back them out to the Gulf of Alaska. The magnitude of these pools is a function of input from escapement, catch and movement between districts and output to the gulf. The daily movement of salmon to the fishery from the gulf is modeled as a fraction of the pool of

fish in their district of entry (gateway).

A stock's initial abundance is the sum of its total escapement and its stock-specific catch contributions. The initial abundance of all wild pink salmon stocks is the sum of the stock abundances.

The parameters of the model are: district-specific catchabilities, gateway-residence times and stock-specific migration matrices. The lack of stock identification information prevents us from distinguishing between removals due to catch and removals due to migration. Thus the catchabilities and the movement rates cannot be estimated simultaneously. For this reason, the migration matrices are assumed to be known and the model estimates the catchabilities and the gateway-residence times which are constrained by total run size. The model is programmed in Fortran and parameter values are estimated by minimizing the sum of squared residuals using a non-linear least squares routine in IMSL.



Intertidal/Supratidal Site Selection Utilizing A Geographic Information System

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A goal of the Coastal Habitat Injury Assessment (CHIA) is to quantify injuries to biological resources inhabiting the intertidal zone throughout the Exxon Valdez oil spill area. The CHIA was designed to provide information for the Natural Resources Damage Assessment program. A Geographic Information System (GIS) was employed by the CHIA to identify candidate study sites. A GIS was used for the following reasons: (1) the large amount of shoreline (potentially in excess of 1,244 miles; 2,002 kilometers) contaminated by the spill, (2) the extreme heterogeneity of shoreline types and degrees of oiling in the spill-affected area, (3) the need to embody the various shoreline habitats and degrees of oiling in the study, and (4) to allow for extrapolation of injury determinations to the universe of all sites in the spill-affected area.

Candidate sites were stratified into 15 habitat/oiling categories and randomly drawn from the GEO (GEO 1989) Arc/Info (ESRI) data base with probability proportional to size. Shoreline lengths of sites ranged from 100-600 meters; longer sites had a higher probability of candidate selection. Sites less than 100 m were not selected because they were judged to be too small to allow repeated annual visits with new sample plots over the period of the study. The GIS data layers utilized in GEO for site selection included: (1) the mean high water shoreline digitized from U.S. Geological Sur-

vey 1:63,360 quadrangles in the spill-affected area, (2) the Environmental Sensitivity Index (ESI) maps (Hayes & Ruby 1979; RPI 1983a, 1983b, 1985, 1986) which classified the shoreline into 19 geomorphologic types, and (3) the Oil Spill Impact (OSI) maps (ADEC 1989) which classified the shoreline into five degrees of oiling.

From a universe of 21,362 potential sites encompassing 4,950 miles (9,173 kilometers) of shoreline, 424 candidate sites were drawn and 240 sites were ground-truthed in 1989 to validate GIS-depicted geomorphologic and shoreline oiling information and to determine if they were accessible by the supratidal/intertidal survey crews. Candidate sites represented both oiled (treated) and unoled (control) conditions.

Ground-truthing revealed 119 sites that did not fit the GIS assigned habitat/oiling categories. Discrepancies between the GIS classifications and ground-truthed classifications can be attributed to: (1) the strong dependence on aerial overflight information in the ESI and OSI data bases which was not always consistent with characteristics we observed in the intertidal zone, (2) the lag time of 2 to 4 months between the shoreline oiling data in the OSI data base and our site selection surveys, and (3) a digitizing error which misclassified 20 candidate sites in the fine-textured category. Assessable sites that did not fit the GIS assigned categories were reclassified into

the observed category. The end number of sites varied for each of the categories. When the number of sites was over the target sample size for a category, the desired number was drawn at random without replacement. In the case of the sheltered estuarine category, the number of sites was under the target value.

This methodology maintained the probability based sample; however, extreme variation existed in the strata. For example, the control strata in Prince William Sound included many sites on the mainland with low salinity while oiled strata contained mostly sites on the islands with higher salinity. This problem was rectified during the winter of 1989-1990 by retaining all sites reclassified into the moderately-heavily oiled strata and pairing each with a control site based on physical characteristics and proximity to the oiled site.

Ultimately, a total of 97 study sites (50 treated and 47 controls) were selected for intensive sampling of intertidal biota by the CHIA. Sites were distributed throughout the spill-affected area, consisting of 37 sites in the Prince William Sound area, 27 sites in the Cook Inlet/Kenai Peninsula area, and 33 sites in the Kodiak/Alaska Peninsula area. This study design maintained the probability based inferences to the effects of oil on the intertidal biota in the "adjusted universe" of accessible, moderately-heavily oiled sites greater than 100 m subject to the protocol by which paired control sites were selected.

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Identification of Marbled Murrelet Nesting Habitat in Southcentral Alaska to Guide Restoration Efforts Following the Exxon Valdez Oil Spill

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The Exxon Valdez oil spill zone is an important population center for the marbled murrelet (*Brachyramphus marmoratus*), a small diving seabird. The marbled murrelet suffered substantial injury from the spill (Kuletz, in prep), and its population in Prince William Sound has declined by 67% since surveys in the early 1970s (Laing and Klosiewski in prep).

In Washington, Oregon and California, the marbled murrelet is listed as threatened under the Endangered Species Act. In these states, marbled murrelet nests have only been found in old-growth conifers, and the only remaining populations of marbled murrelets are in waters adjacent to remnant coastal old-growth forests (Marshall 1988).

Because there is evidence that the loss of critical nesting habitat contributed to the decline of marbled murrelets at lower latitudes, protection of nesting habitat has been proposed to assist natural recovery of the murrelet population in the spill zone. However, little is known about the murrelet in Alaska, and until 1990, there had been no effort to study breeding biology of the marbled murrelet in southcentral Alaska. The only known nests at that time were six ground nests that had been found opportunistically (Day et al. 1983) and one tree nest in southeast Alaska (Quinlan and Hughes 1990).

The goal of this project was to describe upland habitat used by murrelets

in the spill zone to guide acquisition of timber rights and for the development of management guidelines to assist murrelet recovery.

The objectives were to: 1) Determine marbled murrelet habitat requirements and develop criteria for documenting occupied nesting habitat within forested portions of the spill zone, and 2) Survey portions of the spill zone to investigate upland murrelet use in the full spectrum of available habitat. During 1991 and 1992, in cooperation with the U.S. Forest Service, we studied murrelet activity during the breeding season throughout western Prince William Sound and described nesting habitat at Naked Island, located in the center of Prince William Sound.

The basic method used to survey murrelets has been the 'dawn watch', whereby observers record murrelet vocalizations and flight patterns around dawn, when murrelets fly to their inland nests (Paton et al. 1990). Because this protocol had not been used in Alaska, we conducted a pilot study in 1990 (Kuletz 1991) and determined that the 'intensive dawn watch' was best suited to our field situation.

We also tested for weather effects and diel (daily) and seasonal variability by conducting intensive dawn watches at three sites above Cabin Bay, Naked Island at bi-monthly intervals from late May to mid August in 1991 and 1992. To locate nests, we used a modified dawn

watch (Naslund et al. 1990), whereby multiple observers are used to pinpoint the suspected nesting site over several mornings. Nest or landing trees were climbed at the end of the season for measurements and collection of nest samples. During all dawn watches we recorded flight behavior and vocalizations to analyze *a posteriori* murrelet activity at sites with known nests, no known nests and flight corridors.

We tested murrelet activity among habitat types in two stages. In 1991, timber type data for Naked, Storey and Peak islands, available in a geographic information system (ARC INFO) from the U.S. Forest Service, were used to describe four forest types based on volume class and stand class, ranging from a low volume, small/young age class to a high volume, large/older age class. We randomly selected 80 sites among the three islands, and a single dawn watch was conducted at 73 of these sites between June and early August, 1991. Forest characteristics were also recorded for each site, and included the diameter at breast height (DBH), height and condition of the 10 nearest overstory trees. The number and type of murrelet detections were used to test for differences in murrelet activity among forest types.

In 1992, dawn watches were conducted at 68 randomly selected sites in western Prince William Sound to determine if there was a non-random distribution of murrelet activity at inland sites. Because timber type coverage was not available, these analyses will be done *a posteriori* among habitat types. The potential for murrelet occupation of non-forested sites will be evaluated with the 1992 data from western Prince William Sound.

At the regularly monitored Cabin Bay

sites, peak murrelet activity occurred 1 hour before official dawn; at lower latitudes, peak activity is at official dawn. On clear days, murrelet activity began earlier, and on mornings with low clouds and fog murrelet activity continued for a longer period after dawn. A peak in murrelet detections occurred between mid and late July, and detections declined abruptly after early August. Thus, in southcentral Alaska dawn watches should begin at least 90 minutes before official dawn and surveys should be conducted from early May to early August. Because of the increase in murrelet activity in late July, survey effort should be distributed evenly throughout the summer among sampling strata, or a weighting factor applied.

On Naked Island in 1991 and 1992, we located seven active marbled murrelet nests, three trees with nest cups, and eleven trees where murrelets landed during the dawn activity period. Based on the characteristics of these 21 trees, marbled murrelets in Prince William Sound appear to nest in old-growth, moss-covered conifers. Analysis of behavioral data among occupied and unoccupied sites will refine the interpretation of behaviors observed at other locations.

Among the 1991 randomly chosen sites on Naked, Peak and Storey islands, we found a positive relationship between high volume, larger age-class forest and murrelet nesting activity. On-site measurements of overstory tree DBH for these sites was also positively correlated with the number of murrelet detections. However, not all high volume forests had murrelet activity. Factors such as slope, aspect, elevation, canopy closure, number and size of moss platforms and other tree and stand characteristics may prove

to be important criteria. Thus, although we can now define potential murrelet nesting habitat, the value of a specific land parcel as murrelet nesting habitat can only be determined by a site-specific survey.

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Homing and Straying Patterns of Coded Wire Tagged Pink Salmon in Prince William Sound.

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Homing and straying patterns of coded-wire tagged pink salmon *Oncorhynchus gorbuscha* originating from four hatcheries and six streams in Prince William Sound were assessed for the 1991 return year. Prince William Sound hatcheries released 615 million pink salmon fry in 1990. Of these, 1,032,000 fry were tagged with coded-wire tags using 32 unique tag codes. Outmigrations from six study streams yielded 2,120,000 wild stock pink salmon fry of which 258,000 were tagged. Coded tags identified a fish release location, its release or outmigration date and, for hatchery bred salmon, the rearing strategy employed prior to release.

Initially, tagged juvenile salmon were recovered during their early marine life and were used to compare growth and survival for salmon from oiled and unoiled areas of Prince William Sound. Tagged adults returning in 1991 were recovered in commercial fisheries allowing managers to assess the contribution of an individual hatchery's or stream's production to the overall commercial catch and to compare the ocean survival of salmon stocks of known oil exposure history.

While enumerating the wild pink salmon escapements to 46 streams in Prince William Sound, over 788,000 spawned out pink salmon carcasses were examined for the presence of a coded-wire tag. In addition, over 90% of the 1991 broodstock collections at all four hatcheries were similarly inspected for the presence of a coded-wire tag. One

hundred and sixteen tagged pink salmon of hatchery origin were recovered in 27 of the 46 streams examined.

Straying pink salmon from Wally H. Noerenberg Hatchery (WHN) on Esther Island comprised 55% (64 tagged fish) of the total number of hatchery strays recovered and were found in 20 of 46 streams examined. Eighteen of these tagged fish from WHN were recovered from a single stream. Pink salmon from Armin F. Koernig Hatchery (AFK) in southwest Prince William Sound comprised 28% (33 tagged fish) of the total numbers of hatchery strays and were recovered in 16 streams. One AFK tagged pink salmon was recovered at Irish Creek located in eastern Prince William Sound approximately 90 miles from AFK hatchery.

Pink salmon from Solomon Gulch Hatchery (SGH) in Valdez comprised 5% (six tagged fish) of the stray hatchery tagged salmon and were recovered in three streams. Pink salmon from Cannery Creek Hatchery (CCH) comprised 11% (13 tagged fish) of the total number of hatchery strays and were recovered in 11 streams. It should be noted that a majority of the tag recovery effort was directed towards streams in the western, oil impacted areas of Prince William Sound close to where wild stock tags were applied in 1990. Therefore streams nearer to both Cannery Creek and Solomon Gulch Hatcheries did not receive comparable tag recovery effort as did streams in oil impacted areas.

In examining broodstock collections

for evidence of straying, 12 tagged pink salmon out of a total of 1241 broodstock recoveries were found to be from a hatchery other than their natal location and 3 were found to be from tagged wild stocks. No AFK tagged pink salmon were recovered at other hatcheries although one WHN tagged fish and one tagged wild fish were recovered at AFK. Three CCH, six SGH and one tagged wild fish were recovered in the broodstock collection at WHN. Conversely, two WHN tagged fish were recovered in the CCH broodstock.

At SGH, one tagged wild fish whose natal stream was over 80 miles away was recovered in the SGH broodstock. A total of 616 tagged pink salmon originating from the six wild stock tagging sites were recovered from streams. Three of the wild stock tagging sites (Loomis Creek, Herring Creek and Hayden Creek) were located in oiled locations and three (Totemoff Creek, Cathead Creek and O'Brien Creek) were in unoiled locations, all in western Prince William Sound. For fish tagged at Loomis Creek, 150 of 163 fish (92%) were recovered in their natal stream.

The remaining 13 fish were recovered at eight different streams located from 1 to 15 miles away from Loomis Creek. Of those fish tagged at Hayden Creek on LaTouche Island, 84 of 94 (89%) were recovered at their natal stream. The remaining ten fish were recovered in seven different streams located between 2 and 20 miles from Hayden Creek. At Herring Creek on Knight Island (the most heavily oiled wild stock tagging site), 54 of 117 tagged fish (46%) were recovered on site. The remaining 63 tags were recovered at 18 different streams from 6 to 25 miles away. Fourteen tagged fish from Herring Bay Creek were recovered

at Loomis Creek and an additional 20 tagged fish from Herring Creek were recovered in a single stream in Eshamy Bay approximately 9 miles away.

At Totemoff Creek on Chenega Island, 108 of 140 tagged fish (77%) were recovered on site. The remaining 32 fish were recovered at 14 different creeks. Two tagged fish from Totemoff Creek were recovered at streams in eastern Prince William Sound some 75 miles from Totemoff Creek. Eleven tagged fish from Totemoff Creek were recovered at a single creek less than 1 mile from Totemoff Creek. At O'Brien Creek, 26 of 29 tagged fish (89%) were recovered on site while the remaining 3 fish were recovered at three different streams located between 4 and 25 miles away. At Cathead Creek, 36 of 73 tagged fish (49%) were recovered on site and the remaining 37 fish were recovered at 16 different creeks located from 3 to 20 miles away.

In Prince William Sound, all hatchery bred pink salmon are tagged at a ratio of approximately 1:560 while the six wild stocks were tagged at ratios ranging from 1:3 to 1:15. By extrapolating the number of hatchery tagged fish recovered in a stream to be representative of their untagged cohorts, then it appears hatchery fish contributed approximately 59% of the total escapement to Loomis Creek; 16% to the Herring Creek escapement; 27% to both the Hayden and O'Brien Creek escapements; 12% of the Totemoff Creek escapement; and 13% of the Cathead Creek escapement. Of all streams examined daily in 1991, Loomis Creek had the greatest number of fish straying into the creek (20 hatchery tagged fish and 16 wild tagged fish). At the same time those fish tagged at Loomis Creek strayed the least of the tagged wild stocks. Conversely, fish tagged at

Cathead Creek strayed more than any other tagged wild stocks while receiving the fewest number of stray hatchery (2) and wild (0) tagged fish. Although our understanding of the magnitude of the straying phenomena is limited geographically to those streams extensively examined in 1991, some patterns in straying are evident. Pink salmon from WHN and AFK hatcheries showed a tendency to stray into streams near the hatcheries and along migration corridors in southwestern Prince William Sound. Of the six stream recoveries of stray tagged-fish from SGH, four were from a remote release near Bligh Island.

The numerous wild stocks in Alaska contain the genetic resources necessary for continued production of salmon under shifting environmental conditions. The Genetic policy of Alaska (Davis, 1985) acknowledges that genetic diversity buffers biological systems against disaster, either natural or human-induced. Maintaining genetic diversity both within and between local populations is essential for the long-term sustained production of Alaska salmon (Mathisen, 1991).

There is evidence that the offspring of hatchery-produced pink salmon may be less viable in the wild than those from local wild fish (Windsor, 1990). Further evidence indicates that rapid expansion of hatchery production coupled with increased exploitation rates usually results in the eventual collapse of the wild stocks (Mathisen, 1991). Increased exploitation of wild stocks by the commercial fishing fleet coupled with the combined effects

of the oil spill and straying by hatchery bred salmon into wild populations has put significant numbers of wild pink salmon populations at risk. For example, wild pink salmon in Prince William Sound, particularly in the oil impacted southwest districts, are unique in that they are predominantly (75%) intertidal spawners, a characteristic which enhances these populations' chances for continued reproductive success especially during harsh winter conditions. However, this characteristic also placed them at great risk for exposure to oil from the Exxon Valdez oil spill and subsequent cleanup activities. Because of the significant amount of straying by hatchery stocks, the potential exists for the eventual displacement of this and other unique adaptations of wild pink salmon populations. The long term productivity of Prince William Sound's wild pink salmon stocks will depend upon the conservation of the genetic diversity among and within those stocks.

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Assessment of Intertidal Algal Populations in Prince William Sound with an Airborne Multispectral Scanner

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One of the goals of the Coastal Habitat Injury Assessment (CHIA) studies is to quantify the damage caused by the *Exxon Valdez* oil spill to intertidal algal populations. The inherent spatial variability of intertidal algal populations suggested that digital remote sensing techniques might be the best way to sample the large geographic area affected by the *Exxon Valdez* spill.

Satellite imagery from the French SPOT satellite has been used to map intertidal algal populations on the Brittany coastline (Moussa et al., 1989). The 20 meter resolution of the SPOT sensor is adequate for this coastline because the topography is fairly flat and the tidal range is over 12 meters which creates a very wide intertidal zone.

The steep, narrow beaches in Prince William Sound would not be adequately mapped by the currently available commercial satellite sensor systems such as Thematic Mapper (TM) and SPOT. A previous study in southeast Alaska (Polcyn et al., 1978) had shown that an airborne scanner could be used to map intertidal algae.

For this study we selected the Compact Airborne Spectrographic Imager (CASI) because the instrument is easily transportable and can be installed on aircraft of opportunity. In addition, processing of the data can be done on microcomputers in the field which greatly assists in ground truthing the imagery. This instrument can achieve a spatial

resolution of 1 square meter per pixel and a spectral resolution of up to 15 spectral bands in its spatial image mode.

The purpose of the study was to determine whether oil induced injuries to *Fucus* and other marine plants could be detected using CASI imagery, and whether these injuries could be quantified and mapped on shorelines throughout the spill area.

The specific objectives were to (1) determine whether the CASI could detect significant differences between the reflectance spectra of *Fucus* and other marine plants in oiled and unoled sites, (2) determine whether the CASI data could be correlated with quantitative data being collected by the CHIA including estimates of percent cover and biomass of *Fucus*, and (3) determine the feasibility of using the CASI to detect and quantify oiled induced injuries to intertidal algal populations throughout the region affected by the *Exxon Valdez* oil spill.

The field work for this study took place in and around Herring Bay on Knight Island, Prince William Sound. Two field visits were made in late July and early August 1990. On the first visit, low clouds and heavy rain prevented flying, but spectral reflectance data was obtained in the laboratory for plant samples taken from six experimental beaches. On the second visit, the weather was again poor, but airborne image data was obtained for 3 pairs of oiled and cleaned (treated) and unoled (control)

beaches, under heavy overcast skies and light rain.

Spectral differences between *Fucus* and *Neorhodomela*, the dominant algal species in the rocky intertidal of Prince William Sound, were found with the CASI in the laboratory study. The unique spectral signature of the marine phanerogam, *Zostera marina*, also allowed this plant to be easily distinguished. Spectral differences were also noted between *Fucus* plants collected on oiled and unoiled beaches. Juvenile plants from oiled beaches were nearly twice as bright (higher reflectance) than those from unoiled sites. Mature plants from the lower intertidal of the oiled site were also brighter than those from the same tidal level at the control site. Mature plants from the upper intertidal levels of the oiled and unoiled sites had similar reflectances.

Based on the spectral differences observed in the laboratory studies, seven spectral bands were chosen for data collection with the airborne CASI. A number of different spectral band combinations were used to quantify the amount of algal biomass on the beaches. These included the "Fluorescence Line Height" algorithm developed by Gower (1978) and Borstad et al (1985) and a number of simple band ratios. A ratio of band 7 (750.5-759.9 nm) to band 6 (665.9-680.6 nm) correlated well with visual estimates of *Fucus* percent cover made in the field ($r^2 = 0.84$).

The total algal cover on the three treated beaches and their paired controls was mapped using the CASI imagery. Because of the low signal levels, algal cover was estimated with a simple supervised classification method that used the relationship between band 7 (750.5 - 759.9 nm) and band 3 (598.4 - 614.7 nm)

and checked visually against a true color composite using bands 3, 2 (538.4 - 561.7 nm), and 1 (478.9 - 500.3 nm). This method did not account for differences in percent vegetative cover in any one pixel, but gave the total area covered by at least some vegetation. The lower threshold of what constituted vegetative was a subjective decision of the operator in this calculation.

Comparison of algal cover showed that the control beaches had a average of 12% greater algal cover than the treated beaches. This algal cover included both *Fucus* and *Neorhodomela*, because the airborne CASI imagery could not differentiate between them. Subtidal *Zostera* was recognized, but no ground data was available to confirm the percent coverage.

The discrimination of different algal species and the different age classes of *Fucus* that had been shown to be possible in the laboratory studies was not possible due to the poor weather conditions at the time of the aerial surveys. The low light levels caused by the heavy overcast and rain caused low signal to noise ratios which hampered spectral analysis. In addition, at this time of year in Herring Bay the intertidal algal species have bleached to a uniform brown color which makes species discrimination by spectral methods difficult.

This study has shown that CASI imagery can be used to quantify intertidal algal populations in Prince William Sound. Low altitude aerial photographs provide higher spatial resolution than the CASI imagery, but have very low spectral resolution, a low dynamic range, and must be redigitized in order to be used in Geographical Information System databases. Future studies of this nature should be carried out in the spring of the year when the different algal spe-

cies could be more easily separated on their spectral characteristics. Digital classification methods should be investigated further, with the goal of obtaining *Fucus* specific indices. This will require more *in situ* data identifying areas of different types of plants and close coordination between biologists and remote sensing specialists. The ability to process image data in the field is very useful in accomplishing this task.

The use of this type of data for mapping large areas of intertidal habitat will require linking Global Positioning System (GPS) navigation data with the CASI imagery. This will allow the data to be referenced to a standard geographic coordinate system and combined with other datasets in the various GIS databases that have been established for this region during the *Exxon Valdez* spill response. Further consideration should also be given to the geometric problems caused by contorted, non-horizontal beach surfaces. It may be possible to acquire aerial photography at the same time as the

CASI imagery and develop digital terrain maps of the beaches onto which the CASI imagery could be superimposed. This would provide better comparison of algal cover between beaches, but would significantly increase the costs of the surveys.

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Stable Carbon Isotope Ratios of Prince William Sound Subtidal Sediments, Prior and Subsequent to the Exxon Valdez Oil Spill

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Numerous investigations have demonstrated the usefulness of stable carbon isotope ratios ($d^{13}C$) of organic matter in sediments and waters in identifying marine regions contaminated with petroleum (e.g., Calder and Parker, 1968; Spies and DesMarais, 1983; Anderson *et al.*, 1983; Eganhouse and Kaplan, 1988). The premise in these investigations was that carbon derived from various organic pools has a characteristic $d^{13}C$ value, e.g., the $d^{13}C$ of terrigenous C3 plants = 25 ‰ (Hong, 1986; Naidu *et al.*, 1992), marine phytodetritus = -21 ‰ (Fry and Sherr, 1984), seagrasses = -10 ‰ (McConnaughey and McRoy, 1979), and Prudhoe Bay crude oil = -30 ‰ (Magoon and Claypool, 1981).

In principle, therefore, the $d^{13}C$ of marine sediments could, based on an isotope mixing equation (Calder and Parker, 1968; Eganhouse and Kaplan, 1988), help to estimate the proportion in the sediment of organic matter derived from various natural or anthropogenic pools. Based on the above premise, we have attempted to examine the possibility of subtidal sediment contamination by Exxon Valdez crude oil in Prince William Sound.

In 1979, 1980 and 1981, 24 prespill sediment samples were collected with a van Veen grab from 37 to 106 m depths throughout Prince William Sound. In 1990 and 1991 two suites of sediment samples, one 'unoiled' and the other 'oiled', were collected from the relatively shallow (<20m) and deep (40-100m) re-

gions of the southwest Sound, using SCUBA divers or a van Veen grab. All samples were stored frozen until ready for analysis. The oiled sites were selected from areas that had adjacent eelgrass and silled fjord habitat shorelines moderately to heavily oiled during the summer and fall of 1989.

The $d^{13}C$ analysis was made on carbonate-free sediments, using a VG 602E mass spectrometer (Naidu *et al.*, in press). The results are expressed relative to the PDB Standard, with a precision of 0.2 ‰. The mean $d^{13}C$ values of the time-series prespill, oiled and unoiled samples were statistically compared using nonparametric tests. Differences between means at $p > 0.05$ were considered insignificant.

The mean $d^{13}C$ of the prespill sediments of 1979 (-22.2 ‰), 1980 (-22.1 ‰) and 1981 (-22.8 ‰) were all similar. The integrated mean $d^{13}C$ values of all the prespill sediments (-22.3 ‰) was similar to the mean $d^{13}C$ values for 1990 shallow and deep oiled sediments (-21.9 ‰ and 22.1 ‰, respectively), and the 1991 shallow and deep oiled sediments (-22.2 ‰ and -22.3 ‰, respectively). No significant differences were detected between the mean $d^{13}C$ of 1990 sediment samples from oiled and control sites of either the shallow or deep-waters. However, in 1991, the mean $d^{13}C$ of shallow (-22.2 ‰) and deep (-22.3 ‰) oiled sediments was lower than that of the shallow (-20.4 ‰) and deep (-21.6 ‰) unoiled sediments.

The finding of similar $d^{13}C$ values in prespill and oiled sediments, from shal-

low and deepwater sites, was contrary to our expectations.

Initially, we postulated that the $\delta^{13}\text{C}$ of prespill sediments in the Sound would be relatively higher (less negative values) than the values for the post-spill oiled sediments. We assumed that any marked contamination of sediments from the Sound with Prudhoe Bay crude oil would shift the $\delta^{13}\text{C}$ of the oiled sediments to more negative values. The discrepancy between our postulation and the analyzed $\delta^{13}\text{C}$ values for prespill and oiled sediments suggests that oiled sediments were not markedly contaminated with oil.

Alternatively, it is possible that petroleum intercalated into the sediments was overwhelmingly diluted by natural organic material (e.g., eelgrass debris). As noted previously, lower $\delta^{13}\text{C}$ values were determined for the 1991 shallow and deep oiled sediments, in comparison with shallow and deep unoiled sediments. It is possible that the source of the lower $\delta^{13}\text{C}$ values in the 1991 sediments is petroleum from the adjacent heavily-oiled beaches. Perhaps sufficient oil had accumulated in the subtidal region by 1991 so that an isotopic signature of oil could finally be detected there. Thus, it appears that at least some oil reworked from the beaches, either by storm waves or tides, is carried offshore and may accumulate in the subtidal region.

In conclusion, we believe that in Prince William Sound sediments, unless heavily contaminated with petroleum, $\delta^{13}\text{C}$ values are of limited use to assess the extent of sediment contamination by crude oil. It is suggested that additional $\delta^{13}\text{C}$ analysis, using GC-IRMS, on the methanol and benzene soluble material (e.g., saturated and aromatic hydrocarbons) of prespill, unoiled and oiled sedi-

ments (Anderson *et al.*, 1983), could provide a more useful index of detecting petroleum contamination of the Prince William Sound sediments than $\delta^{13}\text{C}$ analysis on gross organics of sediments.

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Response to the Exxon Valdez Oil Spill: A New Method to Test the Effects of Residual Oil on Intertidal Recolonization

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Following the Exxon Valdez oil spill, the Alaska Department of Environmental Conservation (ADEC) evaluated the effects of stranded oil on rocky intertidal communities. Field experiments, begun in the winter of 1989, tested the ability of intertidal organisms to colonize oiled substrates. This paper reports on a new field method which paired natural substrates using *in situ* levels of oil. Studies of intertidal communities face the basic problem of variability (Thomas, 1978). In this study, we expected colonization patterns to vary due to the range of intertidal microhabitats encountered as well as differences in surface texture among rocks used as test substrates. A paired design was chosen because of its potential to account for much of this variability (Sokal and Rohlf, 1981) and reduce the need to match control or reference beaches with oiled beaches, a process that can be problematic and can introduce further variance (Mann and Clark, 1978; Thomas, 1978). In this study, pairing was achieved by removing oil from one half of each rock, creating an oiled side and a reference side. Placement of the rocks on an unimpacted beach allowed a controlled examination of the colonization process without the threat of reoiling.

This approach builds on methods used previously to investigate oil spill-related effects on intertidal communities. Previous methods have included:

cleaning substrates (Nelson, 1981; Crapp, 1971), manipulating substrates (Straughan, 1971), and pairing oiled and clean substrates (Straughan, 1971). However, to our knowledge, this is the first method to manipulate natural substrates containing *in situ* levels of oil.

Oiled rocks were gathered from two beaches in Prince William Sound impacted by the stranded oil, one beach on Smith Island (designated as beach segment SM-005 by ADEC) and the second on Eleanor Island (segment EL-057). These beaches were subject to a variety of treatment efforts to remove the oil, including flooding, pressure washing, handwiping, and bioremediation (enhancement of microbial degradation). Despite these removal efforts, which cleaned exposed surfaces, it was easy to obtain oiled rocks for this study by turning rocks over. In mid September, 1989, the Smith Island beach was 70% covered by a 30m wide band of mobile, sticky oil, while at Eleanor Island, although most of the free oil had been removed, rocks were still stained and subsurface oil remained mobile in gravelly areas. The Smith Island beach is a relatively high-energy beach due to its exposure to a long fetch from the north; boulders and large cobbles on the beach are well-rounded, reflecting the impact of storms and wave action. In contrast, the Eleanor Island segment is more sheltered and the rocks are more angular. The unimpacted

site, Gull Island, southeast of the Exxon Valdez grounding site, had a flourishing community of *Fucus*, filamentous algae, barnacles, limpets, snails, mussels, and other organisms typically found in this area (Feder and Bryson-Schwafel, 1988). No stranded oil was observed during any of the visits to this site.

Between September 15 and 18, 1989, rocks were collected from the three beaches, cleaned, and placed back on the beaches in accordance with the following design. Rocks of visually similar levels of oiling were collected. At EL-057, two levels of staining (high and low) were chosen visually. Staining is used here as defined by ADEC (1991) "Oil < 0.1 mm thick, cannot be easily scratched off with fingernail." Stained rocks were cleaned by dipping one half of each rock into methylene chloride. They were then placed randomly into the mid to upper tide area (as judged by the presence of barnacles and other biota) of Gull Island and affixed to the rocky substrate using marine epoxy putty (Z-Spar®). Rocks were retrieved March 17, 1990.

Measurements of algal colonization were made by dividing each rock into two halves corresponding to the oiled and cleaned halves. Five microscope fields were examined on each half; each field was viewed with a grid containing 25 squares of 1 mm². Algal coverage was recorded as the number of squares covered and converted to percent cover.

Using the paired data, following a square-root transformation, algal cover on the cleaned side was regressed against algal cover on the oiled side. The slope, was significantly different from one ($p=0.003$; slope=0.28 after backtransformation), indicating a reduction in percent cover on the oiled sides of about 72%.

The ecological significance of these and other results are reported in detail elsewhere. Here, we describe the statistical utility of the paired design which was evaluated by comparing the results of both nonparametric and parametric tests. All tests indicated a significant ($p<0.05$) difference between percent cover of algae on the two sides of the rocks. The paired t-Test had the lowest p value (0.001). For the unpaired t-Test, p was 0.029, and for the nonparametric tests (Wilcoxon Matched Pairs Test and Mann-Whitney U Test), probabilities were 0.005 and 0.046 respectively. It was not appropriate to conduct a 2-way ANOVA (with site origin and presence of oil as main effects) because one of the sites had zero variance in percent cover on the oiled sides (i.e., there was no algal settlement on the oiled sides of the Smith Island rocks).

The inhibition of algal colonization on oiled sides was very pronounced which probably explains why all of our analyses produced significant results. Our paired design, however, did achieve a greater ability to demonstrate the effect of the residual oil and has been used in subsequent studies.

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The Alaska Heritage Stewardship Program

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The *Exxon Valdez* oil spill brought hundreds of people to the relatively remote beaches of southcentral Alaska for cleanup activities. One effect of this influx was increased awareness of, and access to, archaeological sites which had been protected by their isolation. This awareness prompted an increase in vandalism and damage to sites which has continued since cleanup.

Vandalism is often caused by individuals interested in archaeology but unaware of the damage caused by disturbing sites. Information lost from archaeological sites is irretrievable, and damaged sites cannot be restored to their original condition. Mitigation of such damage generally involves excavation to recover information before further loss occurs. This approach is expensive and time consuming and does not address the ultimate cause of the damage.

Successful archaeological stewardship programs in Arizona, Texas, and Arkansas prevent vandalism through public education and regular patrols of threatened sites (Arizona Site Steward Program, 1992; Texas Archeological Stewardship Network, 1992; Arkansas Archeological Survey, 1992). The Alaska program, focusing on the spill area, is being developed by the U. S. Fish and Wildlife Service and the State Office of History and Archaeology, with help from the Forest Service, National Park Service and the Aluutiq Cultural Center on Kodiak Island.

The program is intended to be self sustaining and locally based. Interested

individuals and organizations will volunteer with participating land managers or owners and receive training. The volunteers will patrol sites, reporting any disturbance and engage in other preservation activities. Governmental involvement will be limited to necessary administrative and record keeping functions, advisory and technical assistance and, if necessary, law enforcement activities.

Public reception has been positive and enthusiastic with interest throughout the spill area as well as the Aleutians, Ketchikan and the Seward Peninsula. Pilot programs are being organized in Kodiak, Homer and Prince William Sound villages and stewards will be active in summer 1993.

The poster shows examples of site vandalism with a brief explanation of the importance of archaeological context. Copies of the Steward Handbook and Fieldbook will be available to illustrate the duties and goals of the program. Photographs of the types of activities stewards will engage in follow. These include site monitoring, collecting oral histories, documenting private artifact collections, and contributing to public education during events like Alaska Archaeology Week.

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Instrumental Neutron Activation Analysis of Nestling Peregrine Falcon Feathers Collected From Prince William Sound and Norton Sound, Alaska, Breeding Populations in 1990

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Secondary remex feathers were collected from nestling Peregrine Falcons (*Falco peregrinus pealei*) in Prince William Sound (n = 32) and Norton Sound (n = 13), Alaska, in 1990. Feather samples were analyzed for trace element content using Instrumental Neutron Activation Analysis.

Concentrations of Sodium, Magnesium, Aluminum, Sulfur, Chlorine, Calcium, Titanium, Vanadium, Manganese, Copper, Iodine, and Nickel were quantified and compared between the two locales and with known concentrations for peregrine breeding populations on Langara Island, British Columbia.

Of primary concern were Vanadium and Nickel concentrations. Both Vanadium and Nickel are considered "signature elements" for Prudhoe Bay crude oil, and concentrations in peregrine feathers may indicate exposure as a result of

the Exxon Valdez oil spill.

Concentrations of Nickel were not present in sufficient quantities to be quantified in the feather samples analyzed from both locales. Concentrations of Vanadium were approximately 1.5 times greater in Prince William Sound samples than in those representing Norton Sound. Overall, trace element concentrations quantified for Prince William Sound were higher than those for Norton Sound and other regions in Alaska.

Concentrations of Chlorine in Prince William Sound feather samples, for instance, were almost 4 times greater than Norton Sound samples, and substantially higher as well than concentrations known for other regions in Alaska. Variation in trace element concentrations was also noted for areas sampled within Prince William Sound as well as Norton Sound.



Hydrocarbons in Intertidal Sediments and Mussels from Prince William Sound, Alaska, 1977-1980: Characterization and Probable Sources

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We collected and analyzed samples of sediments and mussels (*Mytilus trossulus*) for alkane and aromatic hydrocarbons from eight sampling stations adjacent to the oil tanker vessel transportation corridor through Prince William Sound, Alaska, during the period 1977 to 1980, to determine baseline concentrations of these analytes prior to any pollution that might result from oil tanker traffic through the Sound. We evaluated inter-annual (between years) variability of these analytes in sediments and in mussels using two-factor analysis of variance (ANOVA) of logarithm-transformed hydrocarbon concentrations determined in duplicate samples collected in June, 1977 and in June, 1978 at six of the sampling stations.

Intra-annual (within the year) variability was similarly evaluated using ANOVA on results of chemical analyses of duplicate samples collected in May, June, and August, 1978 at seven of the sampling stations. To facilitate comparison with future work, total organic carbon and grain size distribution was determined in the sediment samples, the lipid content was determined in the mussel samples, and the surface seawater temperature and salinity was determined at each sampling station and time.

The results of the hydrocarbon analyses indicate chronic, low-level hydrocarbon contamination that probably originates from small fuel spills, ballast water discharges, and fuel-combustion exhaust

emissions of occasional vessel activity adjacent to three of the sampling stations: Constantine Harbor, Rocky Bay, and Mineral Flats, in decreasing order of contamination, respectively. Contamination at these three stations is indicated by the diversity of aromatic hydrocarbons found in sediments at concentrations that are generally less than 10 ng/g dry sediment weight, but above detection limits (< 1.0 ng/g) of these analytes in sediments.

In contrast, the remaining five sampling stations showed no indication of petroleum hydrocarbon contamination, primarily because few aromatic hydrocarbons were detected at these stations, and detected aromatic hydrocarbons were present only sporadically and at concentrations that were generally near detection limits. Exceptions are perylene, which was found at concentrations well above detection limits at all sampling stations outside Port Valdez, and which probably has natural sources; and phenanthrene, which was found sporadically at all sampling stations and which may also have natural sources, in addition to the hydrocarbon contamination sources indicated at the three polluted stations. Concentrations of aromatic hydrocarbons are too frequently below detection limits at most of the sampling stations to evaluate intra- and inter-annual variability using ANOVA.

Concentrations of individual n-alkanes vary substantially in sediments

and in mussels. The most abundant n-alkanes in sediments include odd carbon-numbered alkanes of molecular weight greater than tetradecane (C-14). Concentrations of these n-alkanes are generally in the range of 10 to 100 ng/g dry sediment weight, and exceed 1000 ng/g at Constantine Harbor. The most abundant n-alkanes in mussels include decane (C-10) through heptadecane (C-17), and pristane, at concentrations generally ranging from 10 to several hundred ng/g dry tissue weight.

Sources of alkanes in sediments include terrigenous plant waxes, marine plankton, and possibly marine macrophytic algae at all the sampling stations, and petroleum-derived alkanes in addition at Constantine Harbor. Terrigenous plant waxes in sediments are indicated by high abundances of odd carbon-numbered n-alkanes of molecular weight greater than nonadecane (C-19) compared with even carbon-numbered n-alkanes in these sediments, and by slight but significant intra-annual variability of these odd carbon-numbered alkanes in sediments, which probably arises from seasonal deposition of senescent leaves. Marine planktonic and algal sources of pristane and normal alkanes is indicated by the presence of these alkanes in sedi-

ments and in mussels, and by the relatively high abundances of pristane, pentadecane (C-15), and heptadecane (C-17) in sediments and in mussels.

Pristane, pentadecane (C-15), and heptadecane (C-17) vary significantly ($P < 0.001$) in sediments, or in mussels, or in both, intra-annually or inter-annually. Pristane variability in sediments and in mussels is significantly correlated, and is probably due to variability of populations of calanoid copepods within and among years in Prince William Sound. Neither pentadecane variability nor heptadecane variability are correlated in sediments and mussels, suggesting multiple biological sources of these alkanes.

These results indicate that, except in areas affected by localized vessel traffic, intertidal sediments and mussels in Prince William Sound are remarkably free of petroleum-contaminant hydrocarbons during the period of this study. The hydrocarbons found in sediments and mussels unaffected by vessel traffic can be adequately explained by known, natural sources. As a result, sediments and mussels contaminated by crude oil from the *Exxon Valdez* oil spill should be particularly apparent, due to the general absence of other confounding sources of petroleum hydrocarbons.



Meiofaunal Recolonization Experiment with Oiled Sediments: Major Meiofauna Taxa

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An in situ experiment was initiated in 1990 in Herring Bay, Prince William Sound, to study the effects of the *Exxon Valdez* oil spill on recolonization by meiobenthos (small, bottom-dwelling organisms). The study site (60°28'0"N latitude, 147°41'12"W longitude) was on a section of shoreline designated as heavily oiled by the Alaska Department of Environmental Conservation; oil remaining from the *Exxon Valdez* spill was obviously present. Temperature and salinity were measured with a self-contained sensing device mounted on a tripod at an elevation of -0.1 m; both remained relatively constant over the first 28 days of the experiment at 8.0±0.1°C and 29.1±0.1 ppt.

Exxon Valdez crude oil was added and mixed into azoic sediments resulting in two concentrations, 0.5% and 1.7% crude oil, and the resulting mixture was added to triplicate colonization trays (all 13 x 28 x 33 cm). In addition, non-oiled azoic sediments treated similarly were added to triplicate trays, and samples were also collected from untreated surrounding sediments to examine treatment effects and the ambient meiofauna community which would probably be the origin of colonizers.

Trays were placed flush with the sediment surface on beaches along a transect paralleling the -0.6 m tidal level, along the upper margins of an eel grass bed. Triplicate samples were collected with

hand-held corers (modified 60 ml plastic syringes) at random locations along X and Y axes within each tray during aerial exposure at low tide on days 0, 1, 2, 29, 90 and 443 after initiation on April 25, 1990.

Cores for meiofauna analysis were preserved in 10% buffered formalin and returned to the laboratory. Hydrocarbon samples were collected with a 3-cm-diameter chrome plated brass tube and placed into hydrocarbon-free glass jars with Teflon lids and frozen until analysis. In the laboratory, meiofauna passing through a 0.500 mesh sieve but retained on a 0.063 mesh sieve were separated from detritus with a sucrose flotation/centrifugation technique (Fleeger, 1979).

All organisms were identified to major taxon with a stereo dissection microscope and enumerated with ruled trays. Predominant taxa (mainly nematodes) were subsampled when they occurred in high densities using a technique which employs a triply-balanced square design (Sherman et al., 1984). Here, we report on the predominant meiofauna taxa from these collections, particularly the nematodes, harpacticoid copepods, copepod nauplii, ostracods and bivalve larvae. Other meiofauna taxa that occurred in substantial numbers on some dates include turbellarians, halocarid mites, gastropod larvae and polychaetes.

Hydrocarbon concentrations in the sediments correlated well with the percent oil added to the treatments. Hydro-

carbon, aromatic and alkane concentrations declined rapidly during the first 30 days of the experiment and became asymptotic. The unoiled sediment treatments were contaminated with very small quantities of hydrocarbons, which also declined over time. Hydrocarbon concentrations in ambient sediments were similar to those in the unoiled treatments.

The experiment was initiated during the late spring, which generally is a period of active meiofauna recruitment in Alaska (Fleeger et al., 1989; Fleeger and Shirley, 1990; McGregor, 1990). Colonization was rapid for many true meiofaunal taxa (but not macrobenthic larvae) which occur in the surface sediments, as densities in trays were not significantly different from surrounding sediment collections by day two, except in the high oil sediments. Generally, high oil treatments had a reduced density compared to low and control sediments until day 29.

After initial colonization, experimental effects independent of treatment were apparent for most taxa. The effects resulted in densities (for most meiofauna taxa) higher in the experimental treatments than densities measured synoptically in the surrounding sediments. Modifications of biotic interactions generated by colonization of an azoic habitat, or emigration/immigration phenomena, may be responsible for the experimental effects.

The type of competitive, agonistic or predator-prey interaction which were altered may explain the variation in magnitude and timing of experimental effects among taxa. The azoic colonization trays may have decreased competition for some taxa, provided others an escape from predation, or influenced both interactions for some taxa. Some predatory

meiofauna may have had altered prey availability, while bacterivorous meiofauna may have experienced increased prey in the oiled sediments.

Harpacticoid copepods are important food items for the early life history stages of many marine fish and crustaceans; because of their importance in marine food webs, they are treated in detail in a separate presentation (Fleeger et al., 1993). In our study, harpacticoids were diverse with > 40 species encountered. Species analysis of the harpacticoid community indicated that sediments and colonization trays were inhabited primarily by phytal copepods associated with the adjacent eel grass and algal mats habitats. We assume the same relationship may have occurred with other surface meiofauna taxa, although they were not identified to species.

The average density of combined live copepods and copepodites were similar in all treatments by day 29, but averaged two to three times higher in experimental treatments than in surrounding natural sediments by day 90. The elevated densities in the experimental treatments persisted on day 443 the following year. The biotic mechanism(s) responsible for the increased densities cannot be determined by our single-factorial experiment. Copepods may have actively selected the experimental trays, or may have had enhanced survival and production after immigration.

Copepods which were assumed to be dead (as determined by deterioration or missing appendages) at the time of collection were counted separately. Dead copepods were present in the sediments used in the experiments as a result of the repeated freezing and thawing technique used to render the sediments azoic, and created some methodological problems.

Dead copepods were present in all samples through day 90, but were higher in experimental trays and rare in ambient sediments. The highest density of dead copepods were found in the high oil treatment on day 29, but few significant differences existed among the treatments throughout the experiment due to high variance.

Decreased availability or active selection against immigration into the experimental trays by copepod nauplii was evident early in the experiment. Nauplii occurred in highest density in the ambient sediments on day 2 (124 ± 79 .core-1), while at the same time nauplii densities in the treatments were extremely low (4.8 ± 5.8 , 7.4 ± 5.2 and 22.3 ± 18.9 in high oil, low oil and unoiled sediments, respectively). Higher densities in the experimental treatments were not found until day 29, when no significant differences occurred among the treatments (142 ± 136 , 103 ± 80 , 84 ± 73 and 74 ± 75 .core-1 for high oil, low oil, unoiled and ambient sediments, respectively).

Nematodes were the numerically predominant taxon on most sampling dates in all treatments, with average densities varying from a minimum of not significantly more than zero at the beginning of the experiment, to a maximum average of 698.core-1 on day 443 in the unoiled trays. As with many other taxa, nematodes had higher average densities in experimental trays in comparison to ambient sediments after day 90, with lowest values in the high oil and highest values in the unoiled treatments.

Ostracods occurred in low average densities in all treatments and colonized rapidly. By day 2, no significant differences occurred among the treatments or ambient sediments. Densities remained low in the high oil treatment through day

443. The highest densities encountered were an order of magnitude higher, in the low oil treatment on day 443, which was also the date of highest density of ostracods in the unoiled treatment and in ambient sediments. Lower densities of ostracods have also been found in some heavily oiled bays in Prince William Sound the year following the oil spill (Shirley et al., 1993).

Halocarid mites, which are often predatory and sometimes predominate in meiofaunal communities in the high intertidal among algae, responded almost identically as the ostracods. They colonized rapidly, had higher densities in the experimental pans than in the ambient sediments, but never attained high densities in any treatment.

Pronounced seasonal changes in density occurred for all taxa in the ambient natural sediments and in the experimental treatments. The seasonal changes varied among taxa in timing and magnitude and reflect seasonal recruitment and mortality events. Natural seasonal and interannual (between years) variation in meiofaunal community composition and density, as is common for most marine metazoans, confound analysis of treatment effects.

Changes in density of temporary meiofauna (the larvae of macrobenthic invertebrates, e.g., bivalves, gastropods and polychaets) occur in pulses related to planktonic settlement in the intertidal zone in Alaska (McGregor, 1990). Bivalve larvae were rare in our cores until day 90, when they were abundant in all treatments and in ambient sediments. Average density of bivalve larvae was 3-4 times higher in the experimental trays than in ambient sediments, suggesting active selection by the larvae or higher post-settlement survival rates.

Our data demonstrate that over small spatial scales, meiofauna recolonize azoic sediments in the intertidal rapidly following an oil spill, but highly oiled sediments reduce recolonization rates of major meiofauna taxa and have effects that are persistent for more than a year for some taxa.

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Responses of Intertidal and Subtidal Meiofauna to the Prince William Sound Oil Spill

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The effects of oil spills on meiofauna (small animals on the ocean floor) are poorly known, even though they are primary food items for many newly settling or postmetamorphic fish and macroinvertebrates. We examined the responses of intertidal (0 m tidal level) and subtidal (-6 m tidal level) meiofauna in Prince William Sound to the *Exxon Valdez* oil spill by comparing community composition and density in oiled and unoiled bays.

Meiofauna were quantitatively sampled in 10 bays within Prince William Sound on five dates in 1989-1990, beginning approximately six weeks after the initial spill. Five bays were unoiled and five received varying amounts of oiling. Samples were collected from five bays on all sampling dates and four bays were sampled on all but one date. Eight samples were collected with hand-held piston corers (modified syringes) to approximately the same sediment depth (2.5 cm) at each tidal level and study site on each date. All subtidal samples and most (72%) of the intertidal samples were collected by SCUBA divers; other samples were collected during low tide exposure. Samples were collected at randomly determined intervals along transects paralleling the shoreline at the selected tidal heights. Divers collected samples in front of their swim path to avoid disturbing surface sediments. A total of 380 cores in 1989 and 304 cores in 1990 were collected

for analysis of meiofauna; an additional 288 cores collected in 1991 are partially analyzed. Additional cores from the transects were collected synoptically for sediment and hydrocarbon analysis. Hydrocarbon samples were collected with a 3 cm-diameter chrome-plated brass tube, placed into hydrocarbon-free glass jars with Teflon lids and frozen until analysis. Cores collected for bacterial activity and abundance along our transects at some sites on selected dates during 1989 for a separate study provide additional correlative information.

Cores for meiofauna analysis were preserved in 10% buffered formalin and returned to the laboratory. Meiofauna passing through a 0.500 mesh sieve but retained on a 0.063 mesh sieve were separated from detritus with a sucrose flotation-centrifugation technique (Fleeger, 1979) and stained with rose bengal. All organisms were identified to major taxon with a stereo dissection microscope and enumerated within ruled trays. Predominant taxa (mainly nematodes) were subsampled when they occurred in high densities using a technique which employs a triply-balanced square design (Sherman et al., 1984).

Here, we report on seasonal changes in density of the predominant meiofauna taxa from oiled and unoiled bays, particularly the nematodes, harpacticoid copepods, copepod nauplii, ostracods and bivalve larvae. Other meiofauna

taxa that occurred in substantial numbers on some dates included turbellarians, halocarid mites, gastropod larvae and polychaets.

Pronounced seasonal changes in density occurred for all taxa, particularly among the temporary meiofauna (larvae of macrobenthic species) reflecting seasonal recruitment patterns which varied among the taxa. Changes in density of temporary meiofauna (the larvae of macrobenthic invertebrates, e.g., bivalves, gastropods and polychaets) occur in pulses related to planktonic settlement in the intertidal zone in Alaska. Their densities often decline markedly over short time intervals due to predation and rapid growth which excludes them from being members of the meiofauna (McGregor, 1990). Our infrequent sampling intervals hinder comparative use of their densities.

Nematodes were consistently the numerically predominant taxon at all sites; their density varied significantly between bays, sampling dates and tidal heights. No consistent trends were obvious in the density of nematodes between oiled and unoiled bays in the intertidal or subtidal. One of the heavily oiled bays, Herring Bay, had the lowest nematode densities in the intertidal zone throughout 1989 in comparison to other bays. Intertidal nematode density in Herring Bay increased to levels not significantly different from other bays by September 1989 and remained at levels comparable to other bays in 1990 and 1991.

Harpacticoid copepods (the sum of adults and copepodites) generally had higher densities in the intertidal than in the subtidal zone for all bays. Our measurements of densities of the entire harpacticoid community are simplistic,

as the harpacticoid community of Prince William Sound is diverse. In related meiofauna recolonization studies conducted in Herring Bay, more than 40 species were identified (Fleeger et al., 1993). Most oiled bays had declines in average copepod density in June, 1989 in comparison to samples collected at the same locations in May; however, the same seasonal decline was evident in several unoiled bays. The lowest densities of harpacticoids measured during the study were in oiled bays, however, some of the highest densities were found in other oiled bays. Most bays, oiled and unoiled, had a trend of declining densities of harpacticoids from April through September in the intertidal, with generally the opposite trend for the subtidal harpacticoids. Similar seasonal cycles have been reported for harpacticoid copepods in both the intertidal (McGregor, 1990) and subtidal in southeastern Alaska (Fleeger et al., 1989; Fleeger and Shirley, 1990).

We know from field experiments in Prince William Sound that some harpacticoid species can recolonize azoic oiled and unoiled sediments quickly (<30 days) over small spatial scales, and that their densities may become higher in oiled sediments than in adjacent, unoiled sediments (Fleeger et al., 1993; Shirley et al., 1993). Similar, but also varying, responses of harpacticoids to oil spills have been reported from other habitats (Fleeger and Chandler, 1983; Decker and Fleeger, 1984).

Ostracods responses were similar to harpacticoid copepods. Several oiled bays (Herring, Iktua, Sleepy) had lower ostracod densities in the intertidal zone than unoiled bays (Eshamy, Ewan, Paddy) during the initial sampling series, while subtidal ostracod densities

were not significantly different among oiled and unoiled bays. A midsummer depression in density was present in all bays in both the intertidal and subtidal; however, increases in ostracod densities occurred in the unoiled bays in the September samples, but generally did not occur in the oiled bays. The same phenomenon was observed for ostracods in recolonization *in situ* experiments, where their densities remained depressed the subsequent year.

In summary, it is probable that the initial depression of the meiofauna community in response to oiling were not measured because of the time lapse (six weeks) between the spill and sampling. The oil spill in Prince William Sound occurred at a time of annual recruitment for many meiofauna species in Alaska (Fleeger et al., 1989; Fleeger and Shirley, 1990; McGregor, 1990) and concentration of many volatile hydrocarbon components decrease rapidly (Shirley et al. 1993). Putative oil effects may have been evident in the initial samples following the spill in some bays, primarily in intertidal samples. Nematode densities were relatively unaffected, although the lowest densities recorded were in oiled bays. Harpacticoid copepods generally had lower densities (with a notable exception) in the intertidal zone in oiled bays on the initial sampling date after the spill. The same relationship was not true in the subtidal, which appeared to have decreased densities of harpacticoids in

some oiled bays on the subsequent sampling date.

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Exxon Valdez
Oil Spill
Symposium

February 2-5, 1993
Anchorage, AK, USA

Program

Exxon Valdez Oil Spill Symposium Schedule

Day	Tues., Feb. 2	Wed., Feb. 3 — Concurrent Sessions		Thurs., Feb. 4 — Concurrent Sessions		Fri., Feb. 5
Morning	Plenary Session 8:00 A.M. Introduction, Keynote, overviews on fate of the oil and cleanup	Session I IA-Fate & Toxicity 8:30 A.M.	Session II IIA-Salmon 8:30 A.M.	Session I IA-Intertidal 8:00 A.M.	Session II IIA-Archaeology /Human Impacts 8:00 A.M. IIB-Fish 10:00 A.M.	A-Marine and Terrestrial Mammals 8:00 A.M.
Lunch Break	11:30 - 1:00 P.M.	11:50-1:20 P.M.	12:10-1:30 P.M.	11:40-1:00 P.M.	12:00-1:20 P.M.	12:00-1:20 P.M.
Afternoon	Plenary Session (con't) Overviews on archaeology, subsistence, intertidal, subtidal, fish, birds, mammals, and restoration Social Hour 6:00 P.M.	IB-Cleanup and Treatment Effects 1:20 P.M. IC-Subtidal 3:20 P.M.	IIB-Birds 1:30 P.M.	IA-Intertidal (con't) 1:00 P.M. IB-Subsistence 1:40 P.M.	IIC-Herring 1:20 P.M.	A-Marine and Terrestrial Mammals (con't) 1:20 P.M. B-Panel Discussion 3:20 P.M.
Adjourn	6:00 P.M.	4:40 P.M.	4:50 P.M.	4:20 P.M.	4:20 P.M.	5:00 P.M.

Exxon Valdez **Oil Spill Symposium**

Program



Sponsored by:

Exxon Valdez Oil Spill Trustee Council
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Tuesday, February 2, 1993

Plenary Session

Dave Gibbons, Moderator

8:30 A.M.

"Hard Aground"

An audio-visual history of the *Exxon Valdez* oil spill, produced by the State of Alaska

Opening Remarks

Governor Walter Hickel, State of Alaska

Introduction

Trustee Council Member

Keynote Address

Richard Murphy, The Cousteau Society

Break

10:00-10:30 A.M.

**So Why Can't Science Tell Us More About the Effects of the
Exxon Valdez Oil Spill?**

Robert B. Spies, Applied Marine Sciences

**Fate of the Oil Spilled From the T/V *Exxon Valdez* in Prince
William Sound, Alaska**

Douglas A. Wolfe¹, M.J. Hameedi¹, J.A. Galt¹, G. Watabayashi¹, Jeffrey W. Short¹, C.E. O'Clair¹, Stanley Rice¹, J. Michel², James R. Payne³, Joan F. Braddock⁴, S. Hanna⁵, and David M. Sale⁶

¹National Oceanic and Atmospheric Administration; ²Research Planning, Inc.;

³Sound Environmental Services Inc.; ⁴University of Alaska Fairbanks; ⁵Sigma Research Corporation; ⁶SnowOtter Environmental Consulting

The *Exxon Valdez* Cleanup: The First Six Months

Vice Admiral Clyde E. Robbins, U.S. Coast Guard

Lunch

11:30-1:00 P.M.

Cultural Resources and the *Exxon Valdez* Oil Spill

Judith E. Bittner, Alaska Department of Natural Resources

Subsistence

James A. Fall, Alaska Department of Fish and Game

4 — Plenary Session, Tuesday, February 2

Overview of Intertidal Processes, Damages, and Recovery

Charles H. Peterson, University of North Carolina

Subtidal Oil Contamination and Biological Impacts

Stanley Rice, Bruce A. Wright, Jeffrey W. Short, and C.E. O'Clair
National Oceanic and Atmospheric Administration

Break

2:20-2:40 P.M.

Summary of Injury to Fish and Shellfish Associated With the Exxon Valdez Oil Spill

Charles P. Meacham and Joseph R. Sullivan
Alaska Department of Fish and Game

How Do You Fix the Loss of Half a Million Birds?

D. Michael Fry, University of California Davis

Effects of the Exxon Valdez Oil Spill on Marine Mammals in Prince William Sound

Kathryn J. Frost¹, Brenda E. Ballachey², Marilyn E. Dahlheim³, and
Thomas R. Loghlin³

¹Alaska Department of Fish and Game; ²U.S. Fish and Wildlife Service;

³National Oceanic and Atmospheric Administration

Restoration Planning Following the Exxon Valdez Oil Spill

John A. Strand, National Oceanic and Atmospheric Administration

Social Hour and Registration for Technical Session

4:00-6:00 P.M.

Wednesday, February 3, 1993

Session I

Registration for Technical Session

8:00 A.M.

Session IA—Fate and Toxicity

Robert B. Spies, Moderator

8:30 A.M.

Tracking Exxon Valdez Oil from Beach to Deep-water Sediments of Prince William Sound, Alaska

Paul R. Carlson and Keith A. Kvenvolden, U.S. Geological Survey

Characterization of Residual Oil in Prince William Sound, Alaska—3.5 Years Later

Paulene O. Roberts¹, Charles B. Henry Jr.¹, Edward B. Overton¹, and Jacqueline Michel²

¹Louisiana State University; ²Research Planning, Inc.

Toxicity of Intertidal and Subtidal Sediments Contaminated by the Exxon Valdez Oil Spill

Douglas A. Wolfe¹, Margaret M. Krahn¹, Ed Casillas¹, K. John Scott², John R. Clayton, Jr.², John Lunz², James R. Payne³, and Timothy A. Thompson²

¹National Oceanic and Atmospheric Administration; ²Science Applications International Corp.; ³Sound Environmental Services, Inc.

Microbial Activity in Sediments Following the T/V Exxon Valdez Oil Spill

Joan F. Braddock¹, Jon E. Lindstrom¹, Thomas R. Yeager¹, Brian T. Rasley¹, Gregory Winter², and Edward J. Brown³

¹University of Alaska Fairbanks; ²Alaska Department of Environmental Conservation; ³University of Northern Iowa

Contamination of Subtidal Sediments by Oil From the Exxon Valdez in Prince William Sound, Alaska

C.E. O'Clair, Jeffrey W. Short, and Stanley Rice

National Oceanic and Atmospheric Administration

Break

10:10-10:30 A.M.

Methods for Determining Crude Oil Contamination in Sediments and Biota After the Exxon Valdez Oil Spill

Margaret M. Krahn, Gina M. Ylitalo, Douglas G. Burrows, Jon Buzitis, Sin-Lam Chan, and Usha Varanasi
National Oceanic and Atmospheric Administration

Nearshore Subtidal Transport of Hydrocarbons and Sediments Following the Exxon Valdez Oil Spill

David M. Sale¹, James Gibeau², and Jeffrey W. Short³
¹SnowOtter Environmental Consulting; ²University of Texas at Austin;
³National Oceanic and Atmospheric Administration

Determination of Petroleum-derived Hydrocarbons in Seawater Following the Exxon Valdez Oil Spill I: Analysis of Seawater Extracts

Jeffrey W. Short and Patricia Rounds, National Oceanic and Atmospheric Administration

Determination of Petroleum-derived Hydrocarbons in Seawater Following the Exxon Valdez Oil Spill II: Analysis of Caged Mussels

Jeffrey W. Short and Patricia Rounds
National Oceanic and Atmospheric Administration

Qualitative and Quantitative Determination of Exxon Valdez Crude Oil in Sediment Samples Using Principal Component Analysis of Hydrocarbon Data

Jeffrey W. Short and Ronald A. Heintz
National Oceanic and Atmospheric Administration

Lunch

12:10-1:30 P.M.

Session IB—Cleanup and Treatment Effects

Pete Peterson, Moderator
1:30 P.M.

Short-term Biological Effects of Shoreline Treatment on Intertidal Biota Exposed to the Exxon Valdez Oil Spill

D.C. Lees¹, W.B. Driskell², and J.P. Houghton³
¹Ogden Environmental and Energy Services Company; ²Consultant; ³Pentec Environmental, Inc.

Growth and Survival of the Predatory Snail *Nucella lamellosa* in Areas Exposed to the Exxon Valdez Oil Spill

T. Ebert¹, D.C. Lees², and H. Cumberland², ¹San Diego State University;
²Ogden Environmental and Energy Services Co. Inc.

Recovery of Prince William Sound Intertidal Infauna from Exxon Valdez Spill and Treatments 1990-1992

J.P. Houghton¹, A.K. Fukuyama¹, W.B. Driskell², D.C. Lees³, Gary Shigenaka⁴, and A.J. Mearns⁴, ¹Pentec Environmental, Inc.; ²Consultant; ³Ogden Environmental and Energy Services Company, Inc.; ⁴National Oceanic and Atmospheric Administration

Recovery of Prince William Sound Intertidal Epibiota from the Exxon Valdez Spill and Treatments—1990-1992

J.P. Houghton¹, A.K. Fukuyama¹, D.C. Lees², W.B. Driskell³, A.J. Mearns⁴, and Gary Shigenaka⁴, ¹Pentec Environmental, Inc.; ²Ogden Environmental and Energy Services Company, Inc.; ³Consultant; ⁴National Oceanic and Atmospheric Administration

NOAA's Long-term Ecological Recovery Monitoring Program: Overview and Implications of Recovery Trends and Treatment Effects

Alan J. Mearns and Gary Shigenaka, National Oceanic and Atmospheric Administration

Break

3:10-3:30

Session IC—Subtidal

Tracy Collier, Moderator

3:30 P.M.

The Effects of the Exxon Valdez Oil Spill on Benthic Invertebrates in Silled Fjords in Prince William Sound

Stephen C. Jewett¹, Thomas A. Dean², and David R. Laur³

¹University of Alaska Fairbanks; ²Coastal Resources Associates, Inc.;

³University of California Santa Barbara

The Effects of the Exxon Valdez Oil Spill on Epibenthic Invertebrates in the Shallow Subtidal

Thomas A. Dean¹ and Stephen C. Jewett²

¹Coastal Resources Associates, Inc.; ²University of Alaska Fairbanks

The Effects of the Exxon Valdez Oil Spill on Eelgrass and Subtidal Algae

Thomas A. Dean¹, Michael S. Stekoll², and Stephen C. Jewett³

¹Coastal Resources Associates, Inc.; ²University of Alaska Southeast;

³University of Alaska Fairbanks

The Effects of the Exxon Valdez Oil Spill on Infaunal Invertebrates in the Eelgrass Habitat of Prince William Sound

Stephen C. Jewett¹ and Thomas A. Dean²

¹University of Alaska Fairbanks; ²Coastal Resources Associates, Inc.

Session II

Session IIA — Salmon

Phillip R. Mundy, Moderator
8:30 A.M.

Assessment of Injury to Pink Salmon Eggs and Fry

Brian G. Bue, Samuel Sharr, Steve D. Moffitt, and Andrew Craig, Alaska
Department of Fish and Game

Cytochrome P450 Induction and Histopathology in Pre-emergent Pink Salmon from Oiled Streams in Prince William Sound, Alaska

Michael Wiedmer¹, Mark J. Fink¹, and John J. Stegeman²

¹Alaska Department of Fish and Game; ²Woods Hole Oceanographic
Institution

Application of Half Length Coded-wire Tags to Wild and Hatchery Pink Salmon Fry in Prince William Sound, 1989-1991

Daniel Sharp, Carol Peckham, and Jodi Smith
Alaska Department of Fish and Game

Impacts of the Exxon Valdez Oil Spill on the Migration, Growth, and Survival of Juvenile Pink Salmon in Prince William Sound

Mark Willette, Alaska Department of Fish and Game

The Impact of the Exxon Valdez Oil Spill on Juvenile Pink and Chum Salmon and Their Prey in Nearshore Marine Habitats

A.C. Wertheimer, A.G. Celewycz, M.G. Carls, and M.V. Sturdevant
National Oceanic and Atmospheric Administration

Break

10:10-10:30 A.M.

Effect of Oil-contaminated Food on the Growth of Juvenile Pink Salmon, *Oncorhynchus gorbuscha*

M.G. Carls¹, L. Holland, M. Larsen¹, J.L. Lum¹, Dorothy Mortenson¹, S.Y. Wang², and A.C. Wertheimer¹

¹National Oceanic and Atmospheric Administration

²University of Southern Mississippi

Coded Wire Tag Recovery Results for Prince William Sound Pink Salmon, 1989-1991

Carol Peckham, Samuel Sharr, and David G. Evans
Alaska Department of Fish and Game

Pink Salmon Spawning Escapement Estimation in Prince William Sound

Samuel Sharr, Daniel Sharp, and Brian G. Bue
Alaska Department of Fish and Game

A Life History Approach to Estimating the Damage to Prince William Sound Pink Salmon from the Exxon Valdez Oil Spill

Harold J. Geiger¹, Brian G. Bue¹, Sam Sharr¹, A.C. Wertheimer², and T. Mark Willette¹.

¹Alaska Department of Fish and Game

²National Oceanic and Atmospheric Administration

Overescapement Impacts of Kenai River Sockeye Salmon

Dana C. Schmidt, Ken Tarbox, Gary Kyle, Bruce King, Linda Brannian, and Jeff Koenings, Alaska Department of Fish and Game

Lunch

12:10-1:20 P.M.

Session IIB — Birds

Michael Fry, Moderator
1:20 P.M.

Effects of the T/V Exxon Valdez Oil Spill on Murres: A Perspective from Observations at Breeding Colonies

D.R. Nysewander, C. Dipple, G.V. Byrd, and E.P. Knudtson
U.S. Fish and Wildlife Service

How Long to Recovery for Murre Populations, and Will Some Colonies Fail to Make the Comeback?

Dennis Heinemann, Manomet Bird Observatory, Inc.

Effects of the Exxon Valdez Oil Spill on Bald Eagles

Timothy D. Bowman and Philip F. Schempf, U.S. Fish and Wildlife Service

Effects of the Exxon Valdez Oil Spill on Pigeon Guillemots (*Cephus columba*) in Prince William Sound, Alaska

K.L. Oakley and K.J. Kuletz, U.S. Fish and Wildlife Service

Quantification of Habitats in Prince William Sound from Landsat Thematic Mapper Satellite Imagery

Richard Podolsky, GAIA Software

Break

3:00-3:20 P.M.

Effects of the *Exxon Valdez* Oil Spill on Marbled Murrelets

K.J. Kuletz, U.S. Fish and Wildlife Service

Acute and Sublethal Effects of the *Exxon Valdez* Oil Spill on Harlequins and Other Seaducks

Samuel M. Patten, Jr., Alaska Department of Fish and Game

Black Oystercatchers in Prince William Sound: Oil Spill Effects on Reproduction and Behavior

Brian E. Sharp¹ and Mary Cody²

¹Ecological Perspectives; ²U.S. Fish and Wildlife Service

Effects of the *Exxon Valdez* Oil Spill on Black-legged Kittiwakes in Prince William Sound

David B. Irons, U.S. Fish and Wildlife Service

Marine Bird Populations of Prince William Sound, Alaska, Before and After the *Exxon Valdez* Oil Spill

K.K. Laing and S.P. Klowiewski, U.S. Fish and Wildlife Service

Thursday, February 4, 1993

Session I

Session IA—Intertidal

Dave Gibbons, Moderator
8:00 A.M.

Estimation of the Exposure Concentration of the Seawater Soluble Fraction of Crude Oil from Mussel Tissue Concentrations

Thomas D. Mehl and Richard M. Kocan, University of Washington

Coastal Habitat Injury Assessment: Intertidal Algal Communities

Michael S. Stekoll¹, Lawrence Deysher², and Zhanyang Guo¹
¹University of Alaska Fairbanks; ²Coastal Resources Associates, Inc.

Influence of the *Exxon Valdez* Oil Spill on Intertidal Algae: Tests of the Effect of Residual Oil on Algal Colonization

P. Bruce Duncan¹, Anthony J. Hooten² and Raymond C. Highsmith³
¹U.S. Environmental Protection Agency; ²Coastal Resources Associates, Inc. and University of Alaska Fairbanks; ³University of Alaska Fairbanks

Damage and Recovery Rates of *Fucus* in Herring Bay, Knight Island

Peter G. van Tamelen and Michael S. Stekoll, University of Alaska Fairbanks

Meiofaunal Recolonization Experiment with Oiled Sediments: The Harpacticoid Copepod Assemblage

J.W. Fleeger¹, M.A. Todaro¹, T.C. Shirley², and M.G. Carls³
¹Louisiana State University; ²University of Alaska Fairbanks
³National Oceanic and Atmospheric Administration

Break

9:40-10:00 A.M.

Impact of the *Exxon Valdez* Oil Spill on Intertidal Invertebrates Throughout the Oil Spill Region

Raymond C. Highsmith¹, Susan M. Saupe¹, Kenneth O. Coyle¹, Tama Rucker¹, and Wallace Erickson²
¹University of Alaska Fairbanks; ²Western EcoSystems Technology, Inc.

Impacts to Intertidal Invertebrates in Herring Bay, Prince William Sound, Following the Exxon Valdez Oil Spill

Anthony J. Hooten¹ and Raymond C. Highsmith²

¹Coastal Resources Associates, Inc. and University of Alaska Fairbanks; ²University of Alaska Fairbanks

Exxon Valdez Oil Spill: Recruitment on Oiled and Non-oiled Substrates

Anthony J. Hooten¹ and Raymond C. Highsmith², ¹Coastal Resources Associates, Inc. and University of Alaska Fairbanks; ²University of Alaska Fairbanks

Bioavailability of Residual PAHs From the Exxon Valdez Oil Spill

Gary Shigenaka¹ and Charles B. Henry, Jr.²

¹National Oceanic and Atmospheric Administration; ²Louisiana State University

Oiled Mussel Beds in Prince William Sound Two and Three Years After the Exxon Valdez Oil Spill

Malin Babcock¹, Gail Irvine², Stanley Rice¹, Patricia Rounds¹, Joel Cusick², and Christine C. Brodersen¹

¹National Oceanic and Atmospheric Administration; ²National Park Service

Variability of Exxon Valdez Hydrocarbon Concentrations in Mussel Bed Sediments

Patricia Rounds, Stanley Rice, Malin M. Babcock, and Christine C. Brodersen, National Oceanic and Atmospheric Administration

Lunch

12:00-1:40 P.M.

Session IB—Subsistence

Usha Varanasi, Moderator

1:40 P.M.

The Response Process: The Goals of the Oil Spill Health Task Force

Thomas S. Nighswander, Alaska Native Medical Center

Subsistence Uses of Fish and Wildlife Resources in Areas Affected by the Exxon Valdez Oil Spill

James A. Fall, Alaska Department of Fish and Game

Overview of Subsistence Food Safety Testing Program

L. Jay Field, National Oceanic and Atmospheric Administration

Assessment of Exposure of Subsistence Fish Species to Aromatic Compounds Following the Exxon Valdez Oil Spill

John E. Stein, Tom Hom, Catherine A. Wigren, Karen L. Tilbury, Sin-Lam Chan, and Usha Varanasi, National Oceanic and Atmospheric Administration

Break

3:00-3:20 P.M.

Petroleum Hydrocarbons in Alaskan Invertebrate Subsistence Foods Following the 1989 Exxon Valdez Oil Spill

Donald W. Brown, Catherine A. Wigren, Douglas G. Burrows, Karen L. Tilbury, Ronald W. Pearce, Jennie L. Bolton, Susan M. Pierce, L. Jay Field, Sin-Lam Chan, and Usha Varanasi
National Oceanic and Atmospheric Administration

Hazard and Risk Assessment of Crude Oil Contaminants in Subsistence Seafood Samples from Prince William Sound

Michael Bolger, Sara H. Henry, and Clark D. Carrington, U.S. Food and Drug Administration

Session II

Session IIA—Archeology/Human Impacts

Pamela Bergmann, Moderator

8:00 A.M.

The Impact of the Exxon Valdez Oil Spill on Cultural Resources

Albert A. Dekin, Jr., Binghamton University

Investigations of Crude Oil Contamination in Intertidal Archaeological Sites Around the Gulf of Alaska

Douglas R. Reger, Alaska Department of Natural Resources

Generating Damage Restoration Costs for Archaeological Injuries of the Exxon Valdez Oil Spill

Martin McAllister, Archaeological Resource Investigations

Long-term Social Psychological Impacts of the Exxon Valdez Oil Spill

J. Steven Picou¹ and Duane A. Gill²

¹University of South Alabama; ²Mississippi State University

The Economic Impacts of the Exxon Valdez Oil Spill on Southcentral Alaska's Commercial Fishing Industry

Maurie J. Cohen, University of Pennsylvania

Break

9:40-10:00 A.M.

Session IIB—Fish

Ray Hilborn, Moderator
10:00 A.M.

Detecting Population Impacts from Oil Spills: A Comparison of Methodologies

Ray Hilborn, University of Washington

Coastal Habitat Studies: The Effect of the *Exxon Valdez* Oil Spill on Shallow Subtidal Fishes in Prince William Sound

David R. Laur¹ and Lewis Haldorson²

¹University of California Santa Barbara; ²University of Alaska Juneau

Survey of Oil Exposure and Effects in Subtidal Fish Following the *Exxon Valdez* Oil Spill: 1989-1991

Tracy K. Collier, Margaret M. Krahn, Cheryl A. Krone, Lyndal L. Johnson, Mark S. Myers, Sin-Lam Chan, and Usha Varanasi
National Oceanic and Atmospheric Administration

Impact of Oil Spilled from the *Exxon Valdez* on Survival and Growth of Dolly Varden and Cutthroat Trout in Prince William Sound, Alaska

Kelly R. Hepler, Patricia A. Hansen, and David R. Bernard
Alaska Department of Fish and Game

Assessment of Damage to Demersal Rockfish in Prince William Sound Following the *Exxon Valdez* Oil Spill

Andrew G. Hoffmann, Kelley R. Hepler, and Patricia A. Hansen
Alaska Department of Fish and Game

Histopathologic Analysis of Chronic Effects of the *Exxon Valdez* Oil Spill on Alaska Fisheries

Gary D. Marty, Mark S. Okihiro, and David E. Hinton
University of California Davis

Lunch

12:00-1:20 P.M.

Session IIC—Herring

Steve Fried, Moderator
1:20 P.M.

Effects of the *Exxon Valdez* Oil Spill on Herring Embryos and Larvae: Sublethal Assessments, 1989-1991

Jo Ellen Hose¹, Evelyn Biggs², and Timothy T. Baker²

¹Occidental College; ²Alaska Department of Fish and Game

Herring Embryo Stage Sensitivity to Water Soluble Fraction of Prudhoe Bay Crude Oil

Richard M. Kocan¹, Jo Ellen Hose², and Evelyn D. Biggs³

¹University of Washington; ²Occidental College;

³Alaska Department of Fish and Game

Egg-larval Mortality of Pacific Herring in Prince William Sound, Alaska, After the Exxon Valdez Oil Spill

Michael McGurk¹ and Evelyn D. Biggs²

¹Triton Environmental Consultants, Ltd.

²Alaska Department of Fish and Game

Larval Fish Distribution and Abundance in Prince William Sound and Resurrection Bay During 1989

Brenda L. Norcross and Michele Frandsen, University of Alaska Fairbanks

The Prince William Sound Herring Recruitment Failure of 1989: Oil Spill or Natural Causes?

Fritz C. Funk, David W. Carlile and Timothy T. Baker

Alaska Department of Fish and Game

Break

3:00-3:00 P.M.

Adult Herring Reproductive Impairment Following the Exxon Valdez Oil Spill

Richard M. Kocan¹, Timothy T. Baker², and Evelyn D. Biggs²

¹University of Washington; ²Alaska Department of Fish and Game

Summary of Known Effects of the Exxon Valdez Oil Spill on Herring in Prince William Sound, and Recommendations for Future Inquiries

Evelyn D. Biggs and Timothy T. Baker, Alaska Department of Fish and Game

Friday, February 5, 1993

Session A—Marine and Terrestrial Mammals

8:00 A.M.

Sea Otters

Carol Gorbics, Moderator

Rehabilitating Oiled Sea Otters and Other Fur-bearing Marine Mammals: Lessons from the *Exxon Valdez* Oil Spill

Randall W. Davis¹ and Terrie M. Williams²

¹Texas A&M University; ²NOSC Hawaii Laboratory

The Efficacy of the *Exxon Valdez* Oil Spill Sea Otter Rehabilitation Program and the Possibility of Disease Introduction into Recipient Sea Otter Populations

Charles Monnett and Lisa Mignon Rotterman, Enhydra Research

Clinicopathologic Alterations in Oiled Sea Otters Dying Acutely in Rehabilitation Centers

A.H. Rebar¹, T.P. Lipscomb², R.K. Harris², and B.E. Ballachey³

¹Purdue University; ²Armed Forces Institute of Pathology; ³U.S. Fish and Wildlife Service

Histopathologic Lesions in Sea Otters Exposed to Crude Oil

T.P. Lipscomb¹, R.K. Harris¹, R.B. Moeller¹, J.M. Pletcher¹, R.J. Haebler², and B.E. Ballachey³

¹Armed Forces Institute of Pathology; ²U.S. Environmental Protection Agency; ³U.S. Fish and Wildlife Service

Gross Lesions in Sea Otters Found Dead Following the *Exxon Valdez* Oil Spill

T.P. Lipscomb¹, R.K. Harris¹, and B.E. Ballachey²

¹Armed Forces Institute of Pathology; ²U.S. Fish and Wildlife Service

Age Distributions of Sea Otters Dying in Prince William Sound, Alaska After the *Exxon Valdez* Oil Spill

D.H. Monson and Brenda E. Ballachey, U.S. Fish and Wildlife Service

Break

10:00-10:20

Recovery of Sea Otter Carcasses Following the *Exxon Valdez* Oil Spill

A. Doroff, A.R. DeGange, C. Lensink, B.E. Ballachey, James L. Bodkin, and D. Bruden, U.S. Fish and Wildlife Service

An Intersection Model for Estimating Sea Otter Mortality Following the *Exxon Valdez* Oil Spill

James L. Bodkin and Mark S. Udevitz, U.S. Fish and Wildlife Service

Mortality of Sea Otters in Prince William Sound Following the *Exxon Valdez* Oil Spill

Robert A. Garrott¹, L. Lee Eberhardt², and Douglas M. Burn³

¹University of Wisconsin; ²Battelle Memorial Institute; ³U.S. Fish and Wildlife Service

Hydrocarbon Concentrations in Tissues of Sea Otters Collected Following the *Exxon Valdez* Oil Spill

Dan Mulcahy¹ and B.E. Ballachey²

¹BioVet Services; ²U.S. Fish and Wildlife Service

Health, Reproduction, and Survival of Adult, Dependent, and Weanling Sea Otters in Prince William Sound From October 1989 to December 1991

Lisa Mignon Rotterman and Charles Monnett, Enhydra Research

Lunch

12:00-1:20

Other Mammals

Thomas R. Loughlin, Moderator

Effects of the *Exxon Valdez* Oil Spill on River Otters in Prince William Sound

R. Terry Bowyer¹, J. Ward Testa¹, James B. Faro², and Lawrence K. Duffy¹

¹University of Alaska Fairbanks; ²Alaska Department of Fish and Game

Assessment of Damages to Harbor Seals Caused by the *Exxon Valdez* Oil Spill

Kathryn J. Frost and Lloyd F. Lowry, Alaska Department of Fish and Game

Vital Rates and Pod Structure of Resident Killer Whales Following the *Exxon Valdez* Oil Spill

Craig O. Matkin¹, Marilyn E. Dahlheim², Graeme Ellis³, and Eva Saulitis¹

¹North Gulf Oceanic Society; ²National Oceanic and Atmospheric Administration; ³British Columbia Department of Fisheries and Oceans

Assessment of Injuries to Prince William Sound Killer Whales

Marilyn E. Dahlheim¹, and Craig O. Matkin²

¹National Oceanic and Atmospheric Administration

²North Gulf Oceanic Society

Humpback Whale Abundance and Distribution

Olga von Ziegesar¹ and Marilyn E. Dahlheim²

¹North Gulf Oceanic Society

²National Oceanic and Atmospheric Administration

Break

3:00-3:20

Panel Session

A panel of plenary and technical session moderators
discussing symposium highlights.

3:20-5:00 P.M.

Poster Sessions

Kate Wedemeyer and Christine C. Broderson, Coordinators

Authors will be available for discussion at these times:

Wednesday, February 3

3:00-3:30 P.M.

Thursday, February 4

9:40-10:00 A.M.

Integration of Shoreline Oiling Data Sets

Randy Hall, Alaska Department of Natural Resources

Management of Natural Resource Damage Assessment Samples and Analytical Data

C.A. Manen¹, E. Robinson-Wilson², S. Korn¹, and R.L. Britten²

¹National Oceanic and Atmospheric Administration

²U.S. Fish and Wildlife Service

Surface Modeling of Floating Oil, The 1989 Exxon Valdez Oil Spill, Prince William Sound, Alaska

Dorothy Mortenson, Hans Buchholdt, Richard McMahon, and Randall Hall,
Alaska Department of Natural Resources

Hydrocarbons in Mussels and Subtidal Sediments: Graphical Presentation of Hydrocarbon Analysis Data with Geographic Map Data

Jeffrey W. Short, Ronald A. Heintz, and Scott Feldhausen

National Oceanic and Atmospheric Administration

Hydrocarbons in Intertidal Sediments and Mussels from Prince William Sound, Alaska, 1977-1980: Characterization and Probable Sources

John F. Karinen, Malin W. Babcock, Donald W. Brown, William D. MacLeod Jr., L. Scott Ramos, and Jeffrey W. Short, National Oceanic and Atmospheric Administration

Occupational Exposures From Oil Mist During the Exxon Valdez Spill Cleanup

Carl Reller, Alaska Health Project

The Alaska Heritage Stewardship Program

Debra Corbett¹ and Douglas R. Reger²

¹U.S. Fish and Wildlife Service; ²Alaska Department of Natural Resources

Economic, Social, and Psychological Impact Assessment of the Exxon Valdez Oil Spill

Michael A. Downs, Impact Assessment, Inc.

The Social and Economic Characteristics of Kenai, Alaska, and Some of the Effects of the Exxon Valdez Oil Spill

Lynn A. Robbins, Western Washington University

Pre-spill and Post-spill Concentrations of Hydrocarbons in Sediments and Mussels in Prince William Sound

Malin Babcock, John F. Karinen, Jeffrey W. Short, and Christine C. Brodersen,
National Oceanic and Atmospheric Administration

Assessment of Intertidal Algal Populations in Prince William Sound with an Airborne Multispectral Scanner

Gary Borstad¹, Kimbal A. Sundberg², and Lawrence Deysher³

¹G.A. Borstad Associates Ltd.; ²Alaska Department of Fish and Game;

³Coastal Resources Associates, Inc.

Response to the Exxon Valdez Oil Spill: A New Method to Test the Effects of Residual Oil on Intertidal Recolonization

P. Bruce Duncan¹, Anthony J. Hooten², and Arthur H. Weiner³

¹U.S. Environmental Protection Agency; ²Coastal Resources Associates, Inc.

and University of Alaska Fairbanks; ³Alaska Department of Natural Resources

Stable Carbon Isotope Ratios of Prince William Sound Subtidal Sediments, Prior and Subsequent to the Exxon Valdez Oil Spill

A. Sathy Naidu, Stephen C. Jewett, Howard M. Feder, and Max K. Hoberg
University of Alaska Fairbanks

Meiofaunal Recolonization Experiment with Oiled Sediments: Major Meiofauna Taxa

T.C. Shirley¹, M. Carls², J.W. Fleeger³ and N. Schizas¹

¹University of Alaska Fairbanks; ²National Oceanic and Atmospheric
Administration; ³Louisiana State University

Responses of Intertidal and Subtidal Meiofauna to the Prince William Sound Oil Spill

T.C. Shirley¹, J.W. Fleeger², C.E. O'Clair³ and S. Rice³

¹University of Alaska Fairbanks; ²National Oceanic and Atmospheric
Administration; ³Louisiana State University

Intertidal/Supratidal Site Selection Utilizing a Geographic Information System

Kimbal A. Sundberg¹, Lawrence Deysher², and Lyman McDonald³

¹Alaska Department of Fish and Game; ²Coastal Resources Associates, Inc.;

³Western EcoSystems Technology, Inc.

Homing and Straying Patterns of Coded-wire Tagged Pink Salmon in Prince William Sound

Daniel Sharp, Samuel Sharr, and Carol Peckham
Alaska Department of Fish and Game

A Reconstruction of Pink Salmon Wild Stock Runs in Prince William Sound

William D. Templin, Jeremy S. Collie, and Terrance J. Quinn II
University of Alaska Fairbanks

Identification of Marbled Murrelet Nesting Habitat in Southcentral Alaska to Guide Restoration Efforts Following the Exxon Valdez Oil Spill

K.J. Kuletz, N.L. Naslund, and D.K. Marks
U.S. Fish and Wildlife Service

Instrumental Neutron Activation Analysis of Nesting Peregrine Falcon Feathers Collected from Prince William Sound and Norton Sound, Alaska, Breeding Populations in 1990

Jimmie R. Parrish, Parrish Associates, Inc.

Boat-based Surveys of Sea Otters (*Enhydra lutris*) in Prince William Sound, Alaska

Douglas M. Burn, U.S. Fish and Wildlife Service

Exxon Valdez Trustee Council Members

Michael A. Barton

Regional Forester

Alaska Region, Forest Service
U.S. Department of Agriculture

Charles E. Cole

Attorney General, State of Alaska

Curtis V. McVee

Special Assistant to the Secretary

Office of the Secretary

U.S. Department of the Interior

Steven Pennoyer

Director

Alaska Region

National Marine Fisheries Service

National Oceanic and Atmospheric Administration

Carl L. Rosier

Commissioner

Alaska Department of Fish and Game

John A. Sandor

Commissioner

Alaska Department of Environmental Conservation

Symposium Sponsors

Exxon Valdez Oil Spill Trustee Council

University of Alaska Sea Grant College Program

American Fisheries Society, Alaska Chapter

For further information contact:



Brenda Baxter, Coordinator

University of Alaska Fairbanks

Alaska Sea Grant College Program

Fairbanks, AK 99775-5040 USA

phone: 907/474-7086

fax 907/474-6285

e-mail:

BITNET: FNBRM1@ALASKA

INTERNET:

FNBRM1@ACAD3.ALASKA.EDU

UNIVERSITY OF ALASKA FAIRBANKS
SCHOOL OF FISHERIES AND OCEAN SCIENCES

Exxon Valdez Oil Spill Symposium

Alaska Sea Grant College Program
University of Alaska Fairbanks
Fairbanks, AK 99775-5040 USA

Exxon Valdez Oil Spill Symposium

February 2-5, 1993
Anchorage, AK, USA

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Pre-registration

Preliminary Schedule

Tuesday, February 2, 1993	Wednesday, February 3, 1993	Thursday, February 4, 1993	Friday, February 5, 1993
<i>Plenary Session</i> Introduction Introductory Slide Program Keynote Address Overview Presentations Damage Assessment Clean-up Archaeology Subsistence Intertidal Subtidal Fish Birds Marine and Terrestrial Mammals Restoration Planning Social Hour	<i>Technical Session I</i> Fate and Toxicity of Oil Tracking the Spill Toxicity of Oil in Sediments Effects of Clean-up Occupational Health Intertidal Biota Birds Bald Eagles Murre Marbled Murrelets Black Oystercatchers Storm Petrels Kittiwakes Cormorants Peregrine Falcons Subsistence Impacts Affected Fish and Wildlife Food Safety Testing Hydrocarbons in Subsistence Foods Archaeology Contaminated Sites Impacted Cultural Resources Costs of Archaeological Injuries Human Impacts Social Impacts Economic Impacts	<i>Technical Session II</i> Subtidal Impacts Water Column Contamination Sediments Benthic Effects Microbial Activity Marine and Terrestrial Mammals Sea Otters Rehabilitation Histopathological Effects Post-spill Mortality Hydrocarbon in Tissues Health, Reproduction, and Survival Disease Harbor Seals Killer Whales Humpback Whales River Otters Brown Bears	<i>Technical Session III</i> Fish Resources Pink Salmon Injury Injury to Eggs and Fry Effects on Migration, Growth, and Survival Homing Sockeye Salmon Herring Injury Sublethal Effects Recruitment Failure Reproductive Impairment Dolly Varden and Cutthroat Trout Rockfish Groundfish Panel Discussion Restoration and the Future

Pre-registration form for the Exxon Valdez Oil Spill Symposium

Name (Please print or type)

Affiliation

Address

State or
Province

Zip
Code

Country

☐ \$95 pre-registration fee enclosed ☐ \$110 registration fee will be paid at the symposium ☐ I will NOT be attending but would like information on the proceedings

Make checks or money orders payable to: American Fisheries Society, Alaska Chapter
Mail to: Karen Sager, National Marine Fisheries Service, P.O. Box 210029, Auke Bay, AK 99821

Registration

If you plan to attend only the first day of the symposium, there is no charge. If you plan to attend all of the Symposium and hear the presentations of research results, a pre-registration fee of \$95 will be charged. At the door registration fee will be \$110. A book of extended abstracts of the papers being presented will be available to all who register for the Technical Sessions. To pre-register for the Technical Sessions, return the attached pre-registration form. **Make checks or money orders payable to: The American Fisheries Society, Alaska Chapter.**

Fill out and mail in the registration form so that adequate symposium materials will be available. **Mail registration forms and fees to: Karen Sager, National Marine Fisheries Service, P.O. Box 210029, Auke Bay, AK 99821.**

Hotel Accommodations

Guest room accommodations are available at nearby hotels. Special symposium rates are offered at the following hotels:

Anchorage Hilton Hotel
500 West 3rd Avenue
P.O. Box 100520
Anchorage, AK 99501 USA
\$72.00* single room; \$75.00* double room

phone from anywhere in the U.S.: 800/245-2527
fax: 907/265-7140

Anchorage Westmark Hotel
720 West 5th Avenue
Anchorage, AK 99501 USA
\$68.50* single or double room
phone: 907/276-7676
fax: 907/258-4950

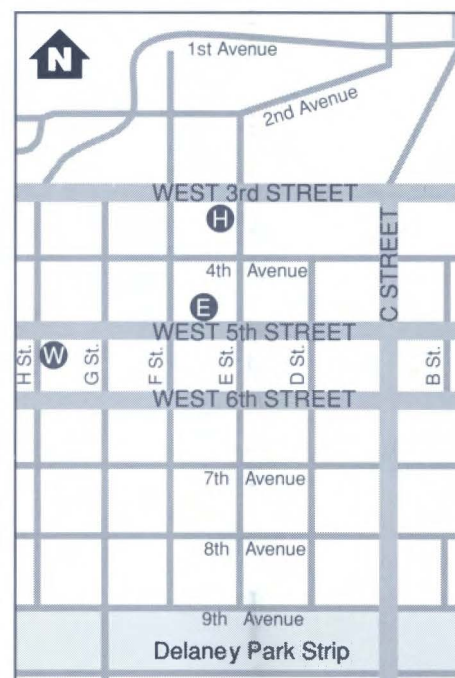
Westmark Reservations Headquarters
phone from anywhere in the U.S.: 800/544-0970
phone from anywhere in Canada: 800/999-2570
fax: 907/258-0560
* plus 8% tax

Hotel reservations should be made directly with the hotel. Be sure to advise them that you will be attending the **Exxon Valdez Oil Spill Symposium** in order to obtain the special rate. Reservations should be made no later than **January 2, 1993.**

Airlines

The following major U.S. airlines have regularly scheduled service to Anchorage International Airport: Alaska Airlines, American Airlines, Delta Airlines, MarkAir, Northwest Airlines, and United Airlines. Several major international air carriers also provide service to Anchorage.

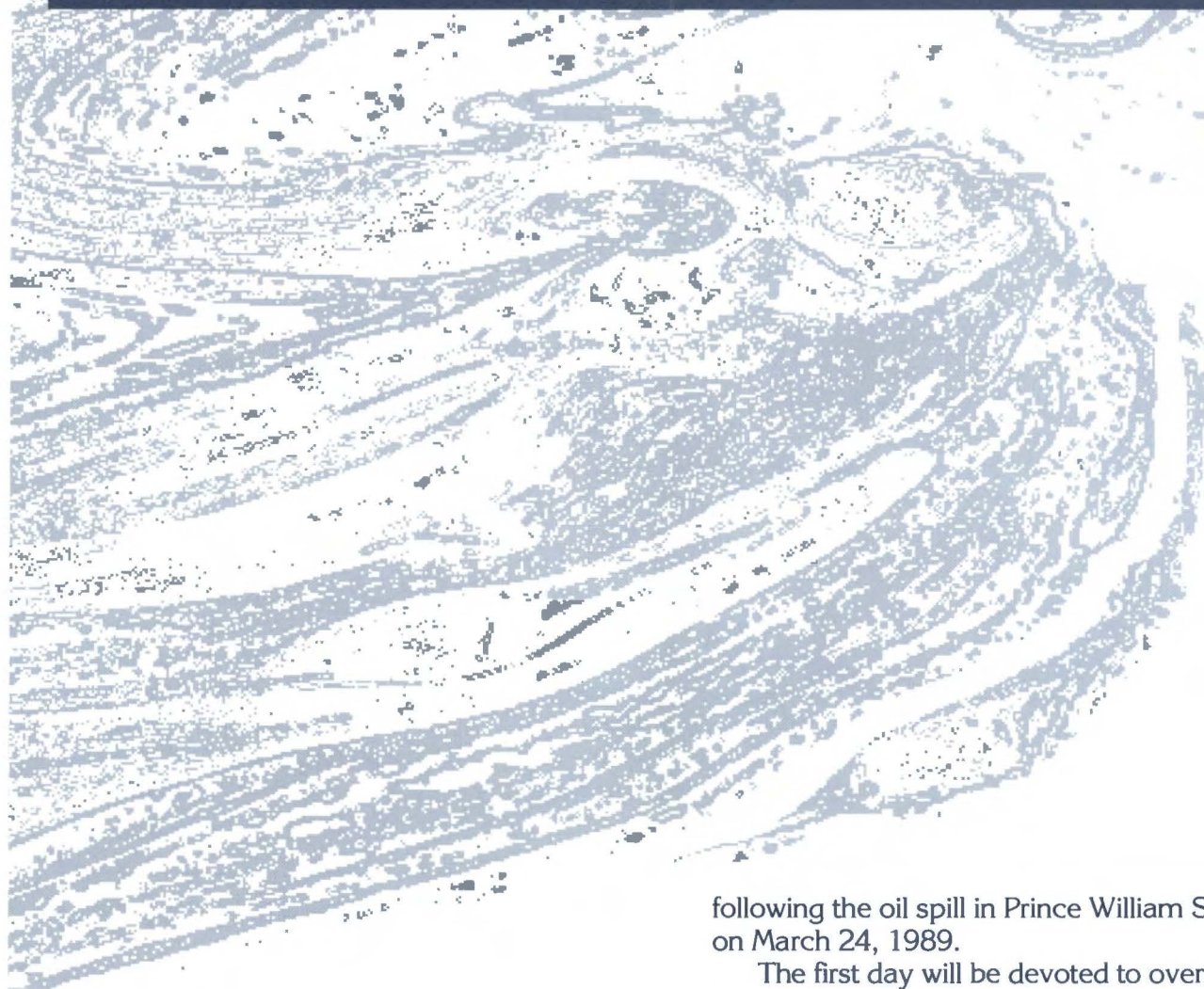
Logistics



- E** Egan Civic and Convention Center
- H** Anchorage Hilton
- W** Anchorage Westmark

Exxon Valdez Oil Spill Symposium

February 2-5, 1993, Anchorage, Alaska



Symposium

The *Exxon Valdez* Oil Spill Symposium is a forum dedicated to presentation of the results of the scientific studies conducted

following the oil spill in Prince William Sound on March 24, 1989.

The first day will be devoted to overview presentations directed toward the general public. Everyone interested in the results of the oil spill research is encouraged to attend. There will be a social hour following the presentations so that attendees can meet

and talk to the *Exxon Valdez* Oil Spill Trustee Council members and scientists who carried out the studies. Presentations will begin at 8:30 AM. There is no charge for attending the first day of the symposium.

The remaining three days (Technical Sessions) will be devoted to presentation of scientific papers and a panel discussion on Restoration and the Future. These presentations will focus on the results of the research conducted for up to four years following the oil spill by the Trustee Council agencies and other organizations. There will be a pre-registration fee of \$95 for the three day Technical Sessions.

A final agenda will be mailed to all registrants prior to the Symposium.

When and Where

The symposium sessions will be held February 2-5, 1993 at the William A. Egan Civic and Convention Center, 555 West 5th Avenue, Anchorage, Alaska.

Proceedings

Proceedings of the *Exxon Valdez* Oil Spill Symposium will be published following the Symposium. Information on ordering the proceedings will be available at the Symposium.

Symposium Sponsors

Exxon Valdez Oil Spill Trustee Council

Members:

Michael A. Barton

U. S. Department of Agriculture

Charles E. Cole

Alaska Attorney General

Curtis V. McVee

U. S. Department of the Interior

Steve Pennoyer

National Oceanic and Atmospheric
Administration

Carl L. Rosier

Alaska Department of Fish and Game

John A. Sandor

Alaska Department of Environmental
Conservation

University of Alaska Fairbanks

Alaska Sea Grant College Program

Exxon Valdez Oil Spill Symposium

Alaska Sea Grant College Program
University of Alaska Fairbanks
Fairbanks, AK 99775-5040 USA

Exxon Valdez Oil Spill Symposium

February 2-5, 1993
Anchorage, AK, USA

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Announcement

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UNIVERSITY OF ALASKA FAIRBANKS
SCHOOL OF FISHERIES AND OCEAN SCIENCES

Exxon Valdez Oil Spill Symposium

February 2-5, 1993 in Anchorage, Alaska, USA

Background

On Friday, March 24, 1989 the 987-foot supertanker *Exxon Valdez* ran aground on Bligh Reef in Alaska's Prince William Sound spilling nearly 11 million gallons of Prudhoe Bay crude oil into one of the world's most productive and beautiful marine environments.

The oil spill occurred just prior to the most biologically active season in southcentral Alaska, exposing plants and animals to the most concentrated, volatile, and potentially damaging forms of the spilled oil. Seaward migrations of salmon fry, major migrations of birds, and the primary reproductive period for most species of birds, mammals, fish, and marine invertebrates took place during the five-month period following the spill.

Coastlines up to 600 miles from Bligh Reef were oiled, including shorelines of Prince William Sound, the Kenai Peninsula, Alaska Peninsula, Kodiak Island, and lower Cook Inlet. These included a number of national parks, national monuments and wildlife refuges, state parks, wilderness parks, marine parks, and a national forest.

In the aftermath of the oil spill, the *Exxon Valdez* Oil Spill Trustee Council was

established to assess natural resource injuries and to plan for restoration. The Trustee Council members oversee many scientific studies to document injuries due to the spill. Until recently the results of these studies were kept confidential due to litigation concerns. The results of the Trustee Council studies are now being released. The symposium sponsors provide this opportunity for the results to be presented to the public.

Symposium

The *Exxon Valdez* Oil Spill Symposium will be a forum for presentation of the results of the scientific studies undertaken following the oil spill in the areas of natural resource damage assessment, response, and restoration. Topics to be considered include injuries to both resources and services related to the *Exxon Valdez* oil spill and associated activities. Participation is not limited by affiliation, but only by scientific merit. Attendance by individuals interested in hearing the presentations and participating in general discussions is encouraged.

The symposium will consist of an opening plenary session giving overviews

of the spill, cleanup, and the resources affected. Technical sessions will follow on the findings of the studies conducted.

Details

The symposium sessions will be held Tuesday through Friday, February 2-5, 1993 at the Egan Convention Center in Anchorage, Alaska. Guest room accommodations at special symposium rates will be available at nearby hotels. A registration fee will be charged. Further information on hotels and symposium registration will be forwarded in November.

For further information, contact:

Brenda Baxter
Symposium Coordinator
University of Alaska Fairbanks
Alaska Sea Grant College Program
Fairbanks, AK 99775-5040 USA

Phone: 907/474-7086
Fax: 907/474-6285
e-mail:
BITNET: FNBRM1@ALASKA
INTERNET: FNBRM1@ACAD3.ALASKA.EDU

Publication Announcement

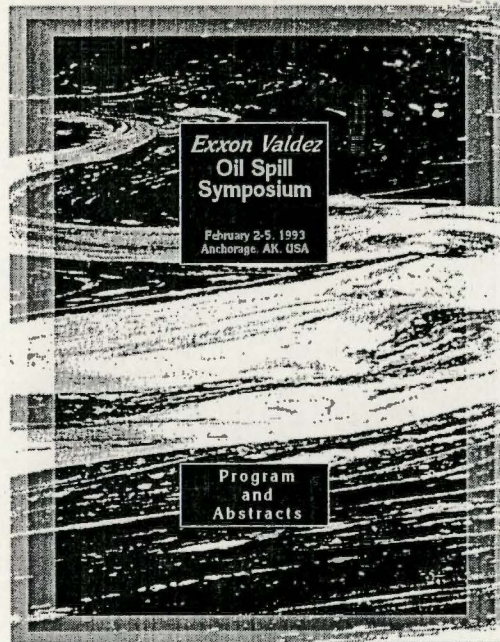
Title Exxon Valdez Oil Spill Symposium: Abstract Book
Anchorage, Alaska, February 2-5, 1993

360 pages

Price \$20.00

Summary

This book contains abstracts of the research presented at the Exxon Valdez Oil Spill Symposium by more than a hundred scientists. According to studies, measurable injury did occur to marine mammals, birds, fish, and shellfish in Prince William Sound. Common murrelets are not mating, having suffered a decrease in population by over 500,000. In 1992, less than half the herring eggs hatched. Oil trapped under mussel beds continues to contaminate wildlife. Papers were also presented on toxicity, subsistence, archeology, and impacts on humans. The symposium was sponsored by the Exxon Valdez Oil Spill Trustee Council, Alaska Sea Grant, and the Alaska Chapter of the American Fisheries Society.



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Communications Department
Alaska Sea Grant College Program
University of Alaska Fairbanks
Fairbanks, AK 99775-5040

Exxon Valdez Oil Spill Symposium: Abstract Book

To:

Title of
Publication

Mailing
Address

Ordering
Information

Quantity: _____ Amount Enclosed: _____
1. Send check, money order, or purchase order payable to the
University of Alaska, OR

2. You can pay with Mastercard or Visa # _____

Expiration date _____ Signature _____

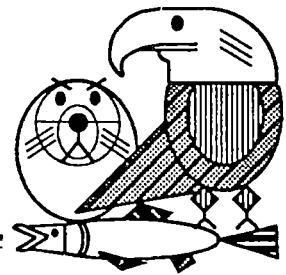


Alaska Sea Grant College Program

University of Alaska, 138 Irving II, Fairbanks, Alaska 99775-5040 (907) 474-7086

Exxon Valdez Oil Spill Trustee Council

645 G Street, Anchorage, Alaska 99501
Phone: (907) 278-8012 Fax: (907) 276-7178



October 8, 1992

Doug Wolfe
NOAA Ocean Assessment Division
WSC-1 Rm 323
6001 Executive Blvd
Rockville, MD 20852

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Dear Doug:

EXXON VALDEZ OIL SPILL
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You are invited to present a plenary session overview at the *Exxon Valdez* Oil Spill Symposium on February 2, 1993, in Anchorage. The symposium, scheduled for February 2-5, will be a forum for presentation of results of injury assessment work undertaken in the wake of the spill. The first day of the symposium will be a plenary session, composed of a keynote speaker, invited overviews, and a social hour. The first day's activities are expressly intended for the general public, and entry will be free of charge.

Technical papers will be presented during the remaining three days of the symposium. Sessions will be organized into the following categories: injury assessment, fate of the oil, clean-up, archaeology, subsistence, intertidal, subtidal, fish, birds, and terrestrial and marine mammals. Attendees during the technical sessions will pay a registration fee.

Because of your experience and work with the *Exxon Valdez* oil spill, the symposium sponsors invite you to give the plenary session presentation "Fate of the Oil." Each overview is scheduled for 20 minutes. A preliminary agenda for the first day of the symposium is enclosed.

A brochure with information on registration and accommodations will be mailed in November. A member of the symposium planning committee will also be contacting you to discuss details by the first week of November.

We hope you will accept this invitation and join us for an interesting and informative symposium. Please respond to either Bruce Wright at 907/789-6601 or me by Friday, October 23.

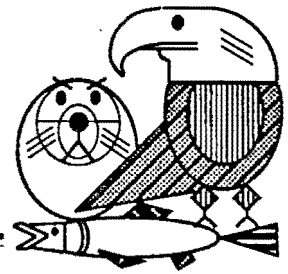
Sincerely yours,
TRUSTEE COUNCIL

By Dave Gibbons, PhD
Interim Administrative Director

cc: Trustee Council

Exxon Valdez Oil Spill Trustee Council

645 G Street, Anchorage, Alaska 99501
Phone: (907) 278-8012 Fax: (907) 276-7178



October 8, 1992

Admiral Clyde Robbins
U.S. Department of Transportation
Office of Intelligence & Security
400 7th Street SW, Room 10401
Washington, D.C. 20590

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EXXON VALDEZ OIL SPILL
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Dear Admiral Robbins:

You are invited to present a plenary session overview at the *Exxon Valdez* Oil Spill Symposium on February 2, 1993, in Anchorage. The symposium, scheduled for February 2-5, will be a forum for presentation of results of injury assessment work undertaken in the wake of the spill. The first day of the symposium will be a plenary session, composed of a keynote speaker, invited overviews, and a social hour. The first day's activities are expressly intended for the general public, and entry will be free of charge.

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Because of your experience and work with the *Exxon Valdez* oil spill, the symposium sponsors invite you to give the plenary session presentation "Cleanup." Each overview is scheduled for 20 minutes. A preliminary agenda for the first day of the symposium is enclosed.

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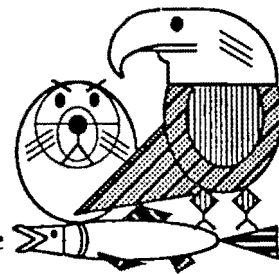
Sincerely yours,
TRUSTEE COUNCIL

By Dave Gibbons, PhD
Interim Administrative Director

cc: Trustee Council

Exxon Valdez Oil Spill Trustee Council

645 G Street, Anchorage, Alaska 99501
Phone: (907) 278-8012 Fax: (907) 276-7178



October 8, 1992

Judy Bittner
Alaska Department of Natural Resources
Office of History and Archaeology
P.O. Box 107001
Anchorage, AK 99510-7001

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Dear Ms. Bittner:

You are invited to present a plenary session overview at the *Exxon Valdez* Oil Spill Symposium on February 2, 1993, in Anchorage. The symposium, scheduled for February 2-5, will be a forum for presentation of results of injury assessment work undertaken in the wake of the spill. The first day of the symposium will be a plenary session, composed of a keynote speaker, invited overviews, and a social hour. The first day's activities are expressly intended for the general public, and entry will be free of charge.

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Because of your experience and work with the *Exxon Valdez* oil spill, the symposium sponsors invite you to give the plenary session presentation "Archaeology." Each overview is scheduled for 20 minutes. A preliminary agenda for the first day of the symposium is enclosed.

A brochure with information on registration and accommodations will be mailed in November. A member of the symposium planning committee will also be contacting you to discuss details by the first week of November.

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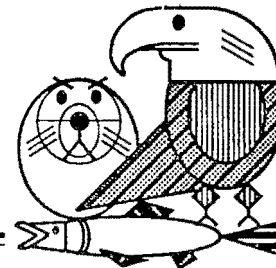
Sincerely yours,
TRUSTEE COUNCIL

By Dave Gibbons, PhD
Interim Administrative Director

cc: Trustee Council

Exxon Valdez Oil Spill Trustee Council

645 G Street, Anchorage, Alaska 99501
Phone: (907) 278-8012 Fax: (907) 276-7178



October 8, 1992

Dr. Jim Fall
Alaska Department of Fish and Game
Subsistence Division
333 Raspberry Road
Anchorage, AK 99518

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EXXON VALDEZ OIL SPILL
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Dear Dr. Fall:

You are invited to present a plenary session overview at the *Exxon Valdez* Oil Spill Symposium on February 2, 1993, in Anchorage. The symposium, scheduled for February 2-5, will be a forum for presentation of results of injury assessment work undertaken in the wake of the spill. The first day of the symposium will be a plenary session, composed of a keynote speaker, invited overviews, and a social hour. The first day's activities are expressly intended for the general public, and entry will be free of charge.

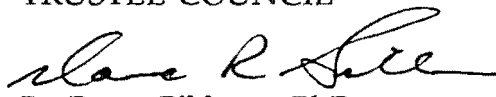
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Because of your experience and work with the *Exxon Valdez* oil spill, the symposium sponsors invite you to give the plenary session presentation "Subsistence." Each overview is scheduled for 20 minutes. A preliminary agenda for the first day of the symposium is enclosed.

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We hope you will accept this invitation and join us for an interesting and informative symposium. Please respond to either Bruce Wright at 907/789-6601 or me by Friday, October 23.

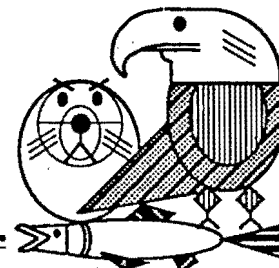
Sincerely yours,
TRUSTEE COUNCIL


By Dave Gibbons, PhD
Interim Administrative Director

cc: Trustee Council

Exxon Valdez Oil Spill Trustee Council

645 G Street, Anchorage, Alaska 99501
Phone: (907) 278-8012 Fax: (907) 276-7178



October 8, 1992

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Dr. Charles Peterson
University of North Carolina
Institute of Marine Sciences
Morehead City, NC 28557

EXXON VALDEZ OIL SPILL
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Dear Dr. Peterson:

You are invited to present a plenary session overview at the *Exxon Valdez* Oil Spill Symposium on February 2, 1993, in Anchorage. The symposium, scheduled for February 2-5, will be a forum for presentation of results of injury assessment work undertaken in the wake of the spill. The first day of the symposium will be a plenary session, composed of a keynote speaker, invited overviews, and a social hour. The first day's activities are expressly intended for the general public, and entry will be free of charge.

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Because of your experience and work with the *Exxon Valdez* oil spill, the symposium sponsors invite you to give the plenary session presentation "Intertidal." Each overview is scheduled for 20 minutes. A preliminary agenda for the first day of the symposium is enclosed.

A brochure with information on registration and accommodations will be mailed in November. A member of the symposium planning committee will also be contacting you to discuss details by the first week of November.

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Sincerely yours,
TRUSTEE COUNCIL

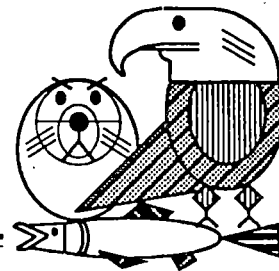
By Dave Gibbons, PhD
Interim Administrative Director

cc: Trustee Council

Exxon Valdez Oil Spill Trustee Council

645 G Street, Anchorage, Alaska 99501

Phone: (907) 278-8012 Fax: (907) 276-7178



October 8, 1992

Dr. Stan Rice
National Marine Fisheries Service
Auke Bay Lab
P.O. Box 210029
Auke Bay, AK 99821

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Dear Dr. Rice:

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ADMINISTRATIVE RECORD

You are invited to present a plenary session overview at the *Exxon Valdez* Oil Spill Symposium on February 2, 1993, in Anchorage. The symposium, scheduled for February 2-5, will be a forum for presentation of results of injury assessment work undertaken in the wake of the spill. The first day of the symposium will be a plenary session, composed of a keynote speaker, invited overviews, and a social hour. The first day's activities are expressly intended for the general public, and entry will be free of charge.

Technical papers will be presented during the remaining three days of the symposium. Sessions will be organized into the following categories: injury assessment, fate of the oil, clean-up, archaeology, subsistence, intertidal, subtidal, fish, birds, and terrestrial and marine mammals. Attendees during the technical sessions will pay a registration fee.

Because of your experience and work with the *Exxon Valdez* oil spill, the symposium sponsors invite you to give the plenary session presentation "Subtidal." Each overview is scheduled for 20 minutes. A preliminary agenda for the first day of the symposium is enclosed.

A brochure with information on registration and accommodations will be mailed in November. A member of the symposium planning committee will also be contacting you to discuss details by the first week of November.

We hope you will accept this invitation and join us for an interesting and informative symposium. Please respond to either Bruce Wright at 907/789-6601 or me by Friday, October 23.

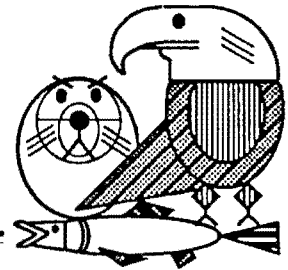
Sincerely yours,
TRUSTEE COUNCIL

By Dave Gibbons, PhD
Interim Administrative Director

cc: Trustee Council

Exxon Valdez Oil Spill Trustee Council

645 G Street, Anchorage, Alaska 99501
Phone: (907) 278-8012 Fax: (907) 276-7178



October 8, 1992

Dr. Robert Spies
Applied Marine Science Inc.
2155 Las Positas Court, Suite S
Livermore, CA 94550

RECEIVED
OCT 14 1992

Dear Bob:

EXXON VALDEZ OIL SPILL
EXXON VALDEZ OIL SPILL
ADMINISTRATIVE RECORD

You are invited to present a plenary session overview at the Exxon Valdez Oil Spill Symposium on February 2, 1993, in Anchorage. The symposium, scheduled for February 2-5, will be a forum for presentation of results of injury assessment work undertaken in the wake of the spill. The first day of the symposium will be a plenary session, composed of a keynote speaker, invited overviews, and a social hour. The first day's activities are expressly intended for the general public, and entry will be free of charge.

Technical papers will be presented during the remaining three days of the symposium. Sessions will be organized into the following categories: injury assessment, fate of the oil, clean-up, archaeology, subsistence, intertidal, subtidal, fish, birds, and terrestrial/marine mammals. Attendees during the technical sessions will pay a registration fee.

Because of your experience and work with the *Exxon Valdez* oil spill, the symposium sponsors invite you to give the plenary session presentation "Damage Assessment Overview." Each overview is scheduled for 20 minutes. A preliminary agenda for the first day of the symposium is enclosed.

A brochure with information on registration and accommodations will be mailed in November. A member of the symposium planning committee will also be contacting you to discuss details by the first week of November.

We hope you will accept this invitation and join us for an interesting and informative symposium. Please respond to either Bruce Wright at 907/789-6605 or me by Friday, October 23.

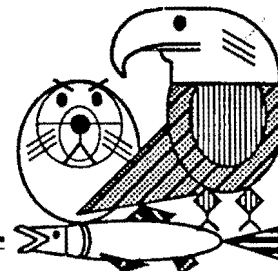
Sincerely yours,
TRUSTEE COUNCIL

By Dave Gibbons, PhD
Interim Administrative Director

cc: Trustee Council

Exxon Valdez Oil Spill Trustee Council

645 G Street, Anchorage, Alaska 99501
Phone: (907) 278-8012 Fax: (907) 276-7178



October 8, 1992

Mr. Sam Sharr
Alaska Department of Fish and Game
P.O. Box 880
Cordova, AK 99754

RECEIVED
OCT 14 1992

EXXON VALDEZ OIL SPILL
TRUSTEE COUNCIL
ADMINISTRATIVE RECORD

Dear Mr. Sharr:

You are invited to present a plenary session overview at the *Exxon Valdez* Oil Spill Symposium on February 2, 1993, in Anchorage. The symposium, scheduled for February 2-5, will be a forum for presentation of results of injury assessment work undertaken in the wake of the spill. The first day of the symposium will be a plenary session, composed of a keynote speaker, invited overviews, and a social hour. The first day's activities are expressly intended for the general public, and entry will be free of charge.

Technical papers will be presented during the remaining three days of the symposium. Sessions will be organized into the following categories: injury assessment, fate of the oil, clean-up, archaeology, subsistence, intertidal, subtidal, fish, birds, and terrestrial and marine mammals. Attendees during the technical sessions will pay a registration fee.

Because of your experience and work with the *Exxon Valdez* oil spill, the symposium sponsors invite you to give the plenary session presentation "Fish." Each overview is scheduled for 20 minutes. A preliminary agenda for the first day of the symposium is enclosed.

A brochure with information on registration and accommodations will be mailed in November. A member of the symposium planning committee will also be contacting you to discuss details by the first week of November.

We hope you will accept this invitation and join us for an interesting and informative symposium. Please respond to either Bruce Wright at 907/789-6601 or me by Friday, October 23.

Sincerely yours,
TRUSTEE COUNCIL

A handwritten signature in dark ink, appearing to read "Dave R. Gibbons".

By Dave Gibbons, PhD
Interim Administrative Director

cc: Trustee Council

Oil Spill Symposium
Progress Report

RECEIVED

SEP 30 1992

At the April 27 Trustee Council meeting, approval was given to organize an Oil Spill Symposium and Proceedings, and the amount of \$25,000 was approved by the Trustee Council for this effort. The Trustee Council instructed the Restoration Team to come back with greater detail on organization, cost estimates, and a timeline at the next Trustee Council meeting. The Trustee Council also instructed the Restoration Team to include non-NRDA sponsored science in the Symposium, and to seek a contractor or co-sponsor (partner) for organizing the Symposium (Alaska Sea Grant was a suggestion).

The Restoration Team is pleased to report that substantial progress has been made on all these items.

1. Symposium Planning Committee

The Restoration Team established a Symposium Planning Committee composed of several Trustee Agency representatives. This committee has and intends to function voluntarily and does not need a support budget. The Committee has met several times to scope out further details of the Symposium planning, and make recommendations to the Restoration Team.

2. Editorial Committee

An Editorial Committee (refer to June 16 memorandum from Dr. Robert Spies) is proposed to deal with the selection of papers for presentation at the Symposium, technical review of Abstracts for the Symposium, and following the Symposium, will arrange peer review of papers for the Proceedings. Costs associated with this effort for this year are included within the existing budgets for the Chief Scientist and for peer review. It is felt that these tasks are within the present scope of duties for reviewing final reports, and no additional funding is required. Within the existing budgets, the estimated costs are \$XX for the Chief Scientist's time, and \$XX for peer reviewers.

3. Symposium Organizer/Co-sponsor

Alaska Sea Grant has expressed genuine interest in as serving as a co-sponsor and organizer for the Symposium (reference attached letters from Brenda Baxter dated June 11 and June 23, 1992). They have indicated that as a co-sponsor, they will assume a portion of the costs of organizing the Symposium. In return, they would become a member of the Symposium Planning Committee. It is the recommendation of the Planning Committee and the Restoration Team that Alaska Sea Grant be asked to co-sponsor the

Symposium, and will be invited to do so immediately.

Who Is Alaska Sea Grant? You have in your package brochures describing Alaska Sea Grants origin and purpose. Briefly, Sea Grant is a federal-state-industry partnership which fosters the wise development, use, and conservation of marine and coastal resources through research, education, and public service. Sea Grant is a national program (based on the Land Grant College concept) that was authorized by Congress in 1966. Alaska Sea Grant was initiated in 1970. Sea Grant is housed within the School of Fisheries and Ocean Sciences at the University of Alaska Fairbanks campus. The fund university-based scientific research projects, education and training projects, and public service projects. Sea Grant is funded by Federal/State matching grants on a \$2 for \$1 basis.

Sea Grant has two major public service activities. One is the Marine Advisory Program with offices currently located in Petersburg, Sitka, Cordova, Kodiak, Homer, and Dillingham. Their other public service function is the Public Information Services project, located in Fairbanks, which provides conference and workshop development and coordination, as well as publications production and distribution, media relations, and special projects.

Since 1975, Sea Grant has coordinated meetings to inform and educate residents of Alaska, the region, and the nation on marine and coastal resource issues. Their objective is to provide expert information on important marine and coastal resources through conferences, workshops, and symposia, and to document these activities in a useful and lasting format.

The expertise of Alaska Sea Grant is ideally suited to co-sponsorship and organization of the Oil Spill Symposium. They have offered to assume the role of organizing the Symposium and in return would receive recognition in part for the scientific and educational value of the Symposium. They are willing to assume some of the costs of the Symposium: i.e., salaries that are required of Sea Grant as co-sponsor and organizer would be Sea Grant's basic contribution and would not be asked for reimbursement.

4. Scope of The Symposium

The Symposium intends to present the scientific results of studies conducted after the oil spill from the areas of natural resource damage assessment, response, and independent science. Participation would not be limited by affiliation, but only by scientific merit. Within existing time scheduling constraints, the scientific presentations of highest merit would be selected first.

5. Symposium Budget Estimates

The following estimates are based on Alaska Sea Grant acceptance as a co-sponsor and contributing partner to the costs of the Symposium. Sea Grant has developed most of the cost estimates for us.

Convention Center Rental (2/2 - 2/5/93) 4 days @ \$1500/day	\$ 6,000
Extended Abstract 1000 copies @ 400 pages	\$13,000
Conference Organization Sea Grant estimates Salaries and Wages	
Coordinator, 1 mo.	\$ 7,200*
Publications Technician, 0.5 mo.	\$ 1,521*
Services - Sea Grant provides** Production of Announcements, Advertisements, Brochures and Programs	\$ 1,275
Printing of Announcements, Brochures, Programs	\$ 1,800
Mailing Costs, 3 mailings @ \$1050 per	\$ 3,150
Telephone	\$ 200
Supplies	\$ 200
Total Estimated Cost of Symposium	\$34,346

* Sea Grant contribution

** Negotiable costs for additional Sea Grant contribution

We estimate attendance at 1,000 registrations. A registration fee of \$35/person should cover all basic costs of the Symposium. The registration fee can be adjusted as further budget detail requires, but we do not anticipate any additional costs that could not be fully supported by a modest registration fee. Additional costs that may accrue include invited travel (to be held to a minimum, if any), and refreshments (we propose to seek a donor for refreshments).

6. Tentative Symposium Schedule

The Symposium is on a tight timeframe. The Symposium will be planned and conducted independent of publishing the Proceedings, but will require coordination. The Proceeding will be prepared after the Symposium, but extended abstracts of presentations and papers will be required for the Symposium. The following tentative schedule is proposed:

Define the scope of symposium	July 1
Identify speakers to be invited	July 15
Produce call for papers	July 20 (mail Aug 5)
Produce Symposium Announcement	July 20 (mail Aug 5)
Invite Speakers	Aug 1
Produce Advertisements	Aug 15
Arrange Catering	Sept
Develop Registration Fee	Sept
Abstracts due	Oct 1
Produce Registration Brochure	Oct 15 (mail Nov 1)
Review Abstracts	Oct 16
Accept Abstracts	Oct 30
Finalize symposium program	Nov 15
Produce abstract book	Jan 4 (to printer)
Produce program brochure	Jan 11 (to printer)
Hold symposium	Feb 2-5

APPLIED RECEIVED
marine
SCIENCES
OIL SPILL OFFICE

JUN 16 1992

June 16, 1992

To: <i>Byron Morris</i>	From: <i>Bob Spies</i>
Co. _____	Co. _____
Dept. _____	Phone # <i>510 373-7142</i>
Fax # <i>407 789-6608</i>	Fax # _____

RECEIVED
JUN 23 1992

To: Restoration Team

From: Robert B. Spies, Chief Scientist; Doug Wolfe, NOAA; Kelley Hepler, ADF&G
EXXON VALDEZ OIL SPILL
NOAA COUNCIL
ADMINISTRATIVE RECORD

Re: Invitations, technical presentations, editing and publication of papers from the *Exxon Valdez* symposium

In anticipation of an agenda item on the symposium for the Trustee Council meeting on June 29th, we are forwarding recommendations relative to a variety of issues on the technical content of the symposium.

1. Formation of an editorial committee. We suggest that we serve on the editorial committee. In addition we would like two more members, perhaps Pete Peterson and one other scientist. This committee would review abstracts, suggest lengths of published papers and arrange the peer review of papers for the proceedings of the symposium.
2. We are pleased with the idea of open invitations. We suggest that everyone who submits an abstract be considered by the committee for presentation of a paper at the symposium and subsequently for publication of a full length paper. We strongly recommend that the Restoration Team commit to having at least one abstract from every significant NRDA study. There were some studies that should not be required to submit an abstract, for example the black bear and deer studies did not document damage. Perhaps the information from some of these studies could be lumped together into one or more short presentations to assure our critics that we are not focusing just on the studies showing damage.
3. We will be pleased to coordinate the invitations to other scientists outside the NRDA process. We would ask the Restoration Team for its recommendations as a preliminary step. When Bob Spies talked briefly to Al Maki, the Chief Scientist for Exxon, he seemed interested in a symposium. We do not know how many of the Exxon Studies would be available, but it is worth making a preliminary enquiry. In addition there are studies sponsored by NOAA HAZMAT, ADF&G and others under response and cleanup.
3. We think that the schedule proposed by Brenda Baxter of Alaska Sea Grant is reasonable.
4. We are proposing that the editorial committee operate as follows:



1. July 1992	assist Alaska Sea Grant in defining topics and identifying speakers to be invited.
2. October 1992	review abstracts submitted, issue acceptances and suggest limits for length of final full-length paper for the proceedings. One short meeting in Anchorage to finalize recommendations.
3. November 1992	assist in finalizing symposium program
4. February 1993	start review of full length papers from symposium
5. May 1993	meet to finalize editorial decisions
6. June 1993	submit papers to publisher

We think it is particularly important that the committee suggest paper length to the authors after the abstract is submitted. This will allow us to estimate the size of the published proceedings--something the publisher will wish to know before finalizing arrangements for publication of the proceedings.

We hope that these recommendations are useful to you in implementing the Trustees decision to hold the symposium. We look forward to assisting the Restoration Team in making this a successful symposium and one which will be of lasting value to the public.

cc: Fraker
Hepler
Wolfe
Broderon
Gibbons
Morris
Rutherford
Bergman
Rice



Alaska Sea Grant College Program

16.1.1

Fairbanks, AK 99775-5040
907-474-7086 • Fax 907-474-6285

University of Alaska Fairbanks
School of Fisheries and Ocean Sciences

June 11, 1992

TO: Byron Morris

FROM: Brenda R. Baxter
Coordinator

SUBJECT: Oil Spill Symposium

RECEIVED

EXXON VALDEZ OIL SPILL
TRUSTEE COUNCIL
ADMINISTRATIVE

Post-It™ brand fax transmittal memo: 7671-1 of pages •

To: <i>Byron Morris</i>	From: <i>Brenda Baxter</i>
Co: <i>NOAA</i>	Co: <i>UAF ASOP</i>
Dept:	Phone: <i>474-6701</i>
Fax: <i>769-6608</i>	Fax: <i>474-6285</i>

Until I have a better understanding of what the Oil Spill Trustees are expecting from the symposium, I will have to guess at what Sea Grant might be able to provide in responding to the symposium needs.

At this time fixed items are that: the symposium will be held in Anchorage, February 1-4, 1993; reports will be presented on the results of scientific studies undertaken because of the Exxon Valdez oil spill; certain individuals will be invited to develop and present papers on defined topics; a call for papers will be circulated to solicit additional papers for presentation; and, all papers will be peer reviewed for publication.

Alaska Sea Grant has been asked to consider participation in the symposium as a co-sponsor and as the coordinator for logistics including: invitations to and all follow up correspondence with invited speakers; preparation and production of a call for papers and other announcements, advertisements, and publicity as deemed necessary; production and publication of abstract book in time for distribution at the symposium; working with caterer to order necessary refreshments for coffee breaks; and development of registration fee to cover identified costs. All of the preceding would be done in concert with the planning committee.

All of the work in organizing the symposium must be completed during 1992. The tasks to be accomplished and a tentative schedule, according to my understanding of the symposium, are:

Defining the symposium	July 1
Defining topics to be covered	July 17
Identifying speakers to be invited	July 31
Production of call for papers*	July 31 (mail Aug 19)
Production of symposium announcement*	July 31 (mail Aug 19)
Invite speakers	Aug 7
Production of print advertisements	Aug 14
Place order for refreshments with vendor	Sept
Development of registration fee	Sept

June 11, 1992
Page Two

Abstracts due	Oct 1
Review of abstracts	Oct 16
Acceptance of abstracts	Oct 30
Production of pre-registration brochure*	Oct 30 (mail Nov 13)
Finalize symposium program	Nov
Production and publication of abstract book	Jan 4 (to printer)
Production and publication of program	Jan 11 (to printer)

* produced and printed by Sea Grant; labeled and mailed by CACI

Estimated budget for the preceding is:

Salaries and Wages (including benefits)

Coordinator 1 mo @ \$7,200.00	7,200.00	} donate by SEA GRANT
Publications Technician 0.5 mo @ \$3042.00	<u>1,521.00</u>	
	8,721.00	

Services

Production of:		\$ 13,950 printing for 400 pages of extended Abstracts for 1000 copies
Graphic concept	150.00	
Call for papers	250.00	
Announcement	250.00	
Advertisements	125.00	
Pre-registration brochure	250.00	
Program and Abstract book covers @ 100	200.00	

Printing of:		
Call for papers (500, 8 1/2" x 11")	300.00	
Announcement (4,000, 8 1/2" x 11")	500.00	
Pre-registration brochure (4,000, 8 1/2" x 11")	500.00	
Program (150, 4" x 9" 12-page plus cover)	500.00	
Abstract book (150, 5 1/2" x 8 1/2" 60-page plus cover)	[1,200.00]	\$ 20,000 for 1500 copies

Communications (phone and postage for contact with speakers)	200.00

Supplies	<u>200.00</u>

TOTAL ESTIMATE	\$13,346.00

June 11, 1992
Page Three

I cannot advise for which of the above costs we will require reimbursement until I know to what extent Sea Grant will participate. Before I can assess the extent of Sea Grant's participation, I will need something in writing outlining the purpose and goals of the symposium, a list of current symposium sponsors, and an idea of which of the above items and others the committee would ask Sea Grant to undertake. As we discussed, if Sea Grant does become a co-sponsor of the symposium, we expect to share in the decision-making and have representation on the planning committee.

Items not addressed are handling and costs of mass mailings, travel arrangements and costs for invited participants, any Sea Grant involvement with the proceedings, and locating hotel accommodations for participants.

I look forward to hearing from you after your committee conference call.

EXXON VALDEZ OIL SPILL SYMPOSIUM REGISTRATIC

RECEIVED
MAR 30 1993

NAME	ADDRESS	AFFILIATION
Adams, Kenneth	PO Box 1855 Cordova, AK	Cordova Dist
Allaway, Jim	2606 East 20th Ave Anchorage, AK 99508	United
Allee, Brian	2353 130th Ave NW Bellevue, WA	Independent
Allen, George	946 E 36th, Rm 603 Anchorage, AK 99508	Harza Northwest
Ammann, Mike	575 Market Rm 3870 San Francisco, CA	Minerals Management Service Chevron Corp.
Andres, Brad	1011 Tudor Rd Anchorage, AK 99503	USF&WS
Armstrong, Sarah	2200 Norwest Center 90 S 7th Street Minneapolis, MN 55402	Faegre & Benson
Astra, Serg A.	PO Box 210888 Auke Bay, AK 99821	DOI/BIA
Babcock, Malin	11305 Glacier Hwy Juneau, AK 99801	NOAA/NMFS
Baker, Tim	333 Raspberry Rd Anchorage, AK 99518	Auke Bay Laboratory Alaska Dept of Fish & Game
Bakus, Prof. Gerald	Univ of Southern CA Los Angeles, CA 90089	Dept. of Biological Sciences
Baldauf, Robert	1849 C St NW #4447 Washington, DC 20240	DOI/Office of Budget
Ballachey, Brenda	1011 E Tudor Road Anchorage, AK 99503	US Fish & Wildlife Ser
Balzar, John PRESS		L.A. Times
Barton, Mike	PO Box 21628 Juneau, AK 99802	US Forest Service
Barter, Paul	403 West 8th Ave. Anchorage, AK 99501	Kinnetic Laboratories, Inc.
Bauer, John	800 Dimond #3-470 Anchorage, AK 99575	ADEC
Bayha, Keith	1011 Tudor Rd Anchorage, AK 99503	USF&WS
Bechtol, William	3298 Douglas St Homer, AK 99603	Alaska Dept of Fish & Game
Beesley, Anne	403 West 8th Ave. Anchorage, AK 99501	Kinnetic Laboratories, Inc.
Bciuce, A. E.	PO Box 2189 Houston, TX 77252	EXXON Production Research
Benda, Bob	PO Box 3470 Valdez, AK	Regional Citizens Advisory Council
Benson, Brian	PO Box 43155 Olympia, WA 98504	Wash State Dept of Fisheries
Benton, Kim	621 W 90th Ave Anchorage, AK 99515	EVOS PAG Forest Products Alternate

Berge, John Arthur	PO Box 69 Korsvoll 0808 Oslo 8, Norway	Norwegian Institute for Water Research
Bergmann, Pamela	1689 C Street Anchorage, AK 99501	DOI/Environmental Affairs Office
Bernard, David	333 Raspberry Road Anchorage, AK 99518	ADF&G
Birnbaum, Molly	PO Box 110030 Juneau, AK 99811	Division of Governmental Coordination
Bittner, Judy	3601 C St #1278 Anchorage, AK 99510	ADNR/Parks & Rec
Bodkin, Jim	1011 E Tudor Rd Anchorage, AK	US Fish & Wildlife Servi
Bocam, Paul D.	Acorn Park Cambridge, MA 02140	Arthur D. Little
Bogle & Gates (10 people)	1031 W 4th Ave #600 Anchorage, AK 99501	Law Offices Of Bogle And Gates
Borresen, Jan Aske	PO 8148DEP 0033, Oslo NORWAY	Ministry of Industry and Energy
Bosche, Dan CREW		ADF&G
Boyer, R. Terry	333 Raspberry Rd Anchorage, AK 99518	ADF&G
Braddock, Joan	Inst. of Arctic Biology Fairbanks, AK 99775	University of Alaska
Brady, James	333 Raspberry Rd Anchorage, AK 99518	Alaska Dept of Fish & Game
Brannian, Linda	333 Raspberry Rd Anchorage, AK 99518	Alaska Dept of Fish & Game
Brendel, Judith	550 W. 7th Anchorage, AK 99501	ARCO
Bridgen, Pamela	11808 Northup Way #104 Seattle, WA 98005	ICF International
Brisson, Lance	3377 Barham Blvd. Los Angeles, CA 91607	Winner/Wagner
Britton, Ronald	1011 E Tudor Rd Anchorage, AK 99503	USF&WS
Brodersen, Mark	410 Willoughby #105 Juneau, AK 99801	ADEC
Bronson, Dr. Michael	201 E. 56th Ave #300 Anchorage, AK 99518	America North/ EMCON, Inc.
Brown Biggs, Evelyn	PO Box 669 Cordova, AK 99574	ADF&G/Commerical Fish Division
Bruce, David	410 Willoughby #105 Juneau, AK 99801	ADEC
Bue, Brian	333 Raspberry Rd Anchorage, AK 99518	Alaska Dept of Fish & Game
Buchholdt, Hans	3601 C Street #916 Anchorage, AK 99503	State of Alaska ADNR
Burns, Bill	P O BOX 2180 Houston, TX 77252	EPR
Bush, James G.	601 Williams Blvd Richland, WA 99352	ICF Kaiser Engineers
Byrd, Vernon	2355 Kachemak Bay Dr Homer, AK 99603	USF&WS
Calkins, Donald	4200 Wilson Blvd #1000 Arlington, VA 22203	National Pollution Funds Center
Carls, Mark G.	11305 Glacier Hwy Juneau, AK 99801	NOAA/NMFS
Carlson, Paul R.	345 Middlefield Rd Menlo Park, CA 94025	Auke Bay Laboratory US Geological Survey

Carlson, Stan	333 Raspberry Rd Anchorage, AK 99518	ADF&G
Carpenter, Keith	PO Box 210855 Auke Bay, AK 99821	USFS
Celewycz, Adrian	11305 Glacier Hwy Juneau, AK 99801	NOAA/NMFS Auke Bay Laboratory
Chalk, Ken	333 Raspberry Rd Anchorage, AK 99518	Alaska Dept of Fish & Game
Channel 2 PRESS	Anchorage, AK	Channel 2 News
Chastain, Brad	1835 S Bragaw St MS538 Anchorage, AK 99512	Alyeska Pipeline Service Company
Chen-Valct, Patty	Marin Headlands, GCNRA Sausalito, CA 94965	The Marine Mammal Center
Chorostecki, Kathleen	7600 Sand Point Way NE Seattle, WA 98115	NOAA/GC/DOJ Damage Assessment
Claybowl, Earl		UAF
Clinton, Dick		Bogla & Gates
Cloud, James L.	PO Box 201014 Anchorage, AK 99520	EVOS/PAG Public-at-Large
Cody, Mary	1011 E Tudor Anchorage, AK 99508	USFWS
Cohen, Maurie	3718 Locust Walk Philadelphia, PA 19104	University of PA Dept of Regional Scien
Cole, Charles	3601 C St. Suite 1210 Anchorage, AK 99503	Trustee Council Off. of Attorney Gener
Collier, Tracy	2725 Montlake Blvd E Seattle, WA 98112	NOAA/NMFS/ECD
Conway, Michael	410 Willoughby #105 Juneau, AK 99801	ADEC/Spill Prevention & Response Division
Cowles, Cleveland	949 E 36th Ave #603 Anchorage, AK 99518	Minerals Management Services
Cox, Bruce A.	PO Box 190168 Anchorage, AK 99519	Marathon Oil Company
Craig, Andy	PO Box 669 Cordova, AK 99574	ADF&G
Crandall, Karen	PO box 25526 Juneau, AK 99802	ADF&G
Cusick, Joel	2525 Gambell Street Anchorage, AK 99503	National Park Service
Cyr, Paul CREW		ADF&G
Dahlheim, Marilyn	7600 Sand Point Way NE Seattle, WA 98115	NOAA/NMFS/NMML
Davis, Steve	4175 Tudor Centre Dr Anchorage, AK 99508	LGL Alaska Research Associates, Inc.
Day, Robert H	PO Box 81934 Fairbanks, AK 99708	Alaska Biological Research, INC
Dean, Dr. Thomas A	1185 Park Center Drive Vista, CA 92083	Coastal Resources Associates, Inc
Dekin, Albert A. Jr	PO Box 6000 Binghamton, NY 13902	Binghamton University
Deysher, Dr. Lawrence	1185 Park Center Drive Vista, CA 92083	Coastal Resources Associates, Inc
Dietrich, Bill PRESS	PO Box 70 Seattle, WA 98111	Seattle Times Newspaper
Dietrick, Larry	410 Willoughby	ADEC

Dillon, Thomas 4 w/No Names	1381 Hillcrest Dr #202 Anchorage, AK 99503	Thomas M Dillon, Attorney at Law
Doudna, David	123 O'Neill Building Fairbanks, AK 99775	UAF/Inst of Marine and Ocean Sciences
Dowling, Gary	PO Box 2180 Houston, TX	EXXON
Downs, Michael A.	2160 Avenida De La Playa La Jolla, CA 92037	Impact Assessment, Inc.
Driskell, William	6536 20th Ave NE Seattle, WA 98115	Consultant
Dubsky, Paul	946 E 36th, Rm 603 Anchorage, AK 99508	Extraordinaire Minerals Management Service
Duffy, Lawrence	333 Raspberry Rd Anchorage, AK 99518	ADF&G
Duffus, Ken	20441 Ptarmigan Blvd Eagle River, AK 99577	Fisherman
Dunbar, Roger	PO Box 669 Cordova, AK 99574	ADF&G
Duncan, P. Bruce	1200 Sixth Ave. Es-098 Seattle, WA 98101	US EPA Region 10
Durda, Judi	9300 Lee Hwy Fairfax, VA	ICF Kaiser Engineers
Ebert, Thomas A.	San Diego St Univ San Diego, CA 92182-0057	Dept of Biology
Ehler, Bud	6001 Executive Blvd Rockville, MD 20852	NOAA/ORCA
Ehlers, Harry R.	PO Box 190168 Anchorage, AK 99519	Marathon Oil Company
Elias, Michael	9300 Lee Hwy Fairfax, VA	ICF Kaiser Engineers
Emerson, Raymond	946 E 36th, Rm 603 Anchorage, AK 99508	Minerals Management Service
Engel, Mary PRESS	1001 Northway Drive Anchorage, AK 99501	Anchorage Daily News
Erickson, Wallace	1402 S Greeley Hwy Cheyenne, WY 82007	West Inc.
Etpison, Suzanne C.	110 Laural Street San Diego, CA 92101	Bixby, Cowan & Gerry Attorneys at Law
Evans, David	333 Raspberry Rd Anchorage, AK 99518	Alaska Dept of Fish & Game
Evans, L.J.	645 G Street Anchorage, AK 99501	ADEC
Evans, Leda	333 Raspberry Rd Anchorage, AK 99518	Alaska Dept of Fish & Game
Evenset, Anita	Strandtorget 2B 9008 Tromsøe NORWAY	Akvaplan Niva AS Norway
Fall, James A.	7260 Huntsman Circle Anchorage, AK 99518	ADF&G
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