Tessera Solar
power from stirling energy systems

January 29, 2010

DOCKET
08-AFC-13
DATE JAN 29 2010
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Mr. Christopher Meyer
CEC Project Manager
Attn: Docket No. 08-AFC-13
California Energy Commission
1516 Ninth Street
Sacramento, CA 95814-5512

Mr. Jim Stobaugh
BLM Project Manager
Attn: Docket No. 08-AFC-13
Bureau of Land Management
P.O. Box 12000
Reno, NV 89520

RE: Calico Solar (Formerly Solar One) Project
Applicant's Responses to Actions Items from the January 5, 2010 Workshop Continuation

Dear Mr. Meyer and Mr. Stobaugh,

Tessera Solar hereby submits the Applicant's Responses to Actions Items from the January 5, 2010 Workshop Continuation. I certify under penalty of perjury that the foregoing is true, correct, and complete to the best of my knowledge.

Sincerely,

Felicia L Bellows

Felicia Bellows
Vice President of Development

Action Items from the Solar One January 5th, 2010 Workshop Continuation

Docket 08-AFC-13

Updated: 1/29/2010

Item Number	Action Item	Responsibility	Response
1	List of items needed from the Applicant or any remaining requests to finish the biological resources analyses (C Huntley mentioned 3 items)	C Huntley	The Applicant has not yet received a list containing remaining items needed. Upon receipt of the list, the Applicant will work as quickly as possible to respond to any necessary items.
2	Docket Hydrologic Assessment	Applicant	Docketed 1/11/2009, CDs arrived 1/12/2009 with hard copies to follow
3	Provide a rationale/discussion on why washes were not treated as a separate vegetation type. If possible, also provide a map showing the location of individuals documented	Applicant	Species found in the "wash habitat" onsite are the same species found in the uplands, with the exception of a few occasional species. These more localized "wash" species were not present in numbers large enough to substantiate a new vegetation category. As a result, washes were not treated as a separate vegetation type. Locations of individual "wash" species were not noted as these species were not sensitive.
4	Provide an idea of the percentage of water conveyance on-site and off-site - what would be conveyed naturally compared to after the Project is built. How much will be flowing in the native system (CFS of percentage)? Will bleeder lines be primary conveyance?	Applicant	The design for the detention basins is to detain 100% of the 100-year event of the off-site flow and meter out 10% of the volume to the existing water course the basin is blocking. The flows generated on site will also be detained in the same manner based upon the stream's velocities at a given point. The intent is to maintain a non-erosive volume throughout the site, if possible.
5	Provide updated figure 12 of the BTR and clarify the proposed site drainage plan post construction	Applicant	The Applicant is working on updating the figure and will submits as soon as it is complete.
6	Provide information on the measured rain fall from 2007 and 2008	Applicant	Winter months range from October to April, winter 2007 = 0.51 inch or 19% of normal, winter 2008 = 2.31 inches or 88% of normal. Average winter rainfall totals is 2.64 inches (3.92 inches for the whole year). From website: http://www.wunderground.com/history/airport/KDAG/2007/5/14/MonthlyHistory.html#calendar. Data not available from website referenced in protocol (http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?cadagg+sca)
7	Feedback on desired survey methods for Burrowing Owl	BLM/CEC/CDFG	The Applicant has begun burrowing owl surveys using the guidance received to date from BLM, CEC, USFWS and CDFG.
8	Provide list of species that may have been mentioned in the Baseline Report but not shown in Figure 3 (C Huntley had referenced a list he had prepared)	CEC	The Applicant has not yet received a list of species requiring clarification. Upon receipt of the list, the Applicant will work as quickly as possible to provide a response.
9	Clarify the presence of those species (in Item 8) upon receipt of that list - clarify whether or not outside of the 10-mile radius	Applicant	Please see the response to Item 8.
10	Map showing the area surveyed in 2009	Applicant	See attached figure provided as Attachment D.
11	Clarify if the Foxtail Cactus was found on Solar One or within the broader "SES Assessment Area"	Applicant	Foxtail cactus (*Coryphantha alversonii*) was detected in the SES Assessment Area, but not within Solar One.

Action Items from the Solar One January 5th, 2010 Workshop Continuation

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Item Number	Action Item	Responsibility	Response
12	Provide a more recent reference (2007 Monitoring Report) for tortoise densities outside of the Project area. Provide supplemental information based on the contents of this report.	Applicant	Tortoise density numbers from the 2007 report show that the West Mojave has a density estimate for 2007 of 4.7 tortoise per square kilometer, this translates into 11.9 tortoise per square mile. East Mojave numbers are 5.8 DT/sq. km which equals 15.0 DT/sq. mile. Citation: U.S. Fish and Wildlife Service. 2009. Range-wide Monitoring of the Mojave Population of the Desert Tortoise: 2007 Annual Report. Report by the Desert Tortoise Recovery Office, U.S. Fish and Wildlife Service, Reno, Nevada.
13	Provide additional information on the Swainson's Hawk and Golden Eagle Detections (dates and locations)	Applicant	Both species were observed as fly-overs only. The specific locations and dates for golden eagle are unavailable, however the golden eagle was observed in 2007 and 2008, while the Swainson's hawk (2 individuals) was only observed March 30, 2008 in the eastern third of the ACEC portion of the survey area. BLM staff have provided location information for golden eagle nest sites. Three nest sites are located within 2-5 miles of the site in the Cady Mountains. The Applicant is currently working with the BLM, CEC, USFWS, and CDFG to determine appropriate survey protocol. The Applicant, at minimum, will survey approximately one mile from the site boundary into the Cady Mountains.
14	Provide a citation for the statement that American Badgers are active during the day	Applicant	1) Long, Charles A. 1973. Taxidea taxus. Mammalian Species. 26: 1-4. 2) http://www.dfg.ca.gov/biogeodata/cwhr/cawildlife.aspx, select American Badger from mammal drop down list and select "Life History"
15	Verify if there are more recent BHS metapopulation surveys/data and if so, provide them	Applicant	Dr. John Wehausen mentioned that recent aerial surveys suggest that the Cady Mountain herd may exceed 300 individuals. CDFG surveyed about 60% of the Cady Mt. sheep habitat in October 2007 and detected 109 individuals.
16	Provide citation and PDF of the Epps et al study related to BHS movement corridors	Applicant	The PDF is provided as Attachment A. The citation is as follows: Epps, C. W., J. D. Wehausen, V. C. Bleich, S. G. Torres, and J. S. Brashares. 2007. Optimizing dispersal and corridor models using landscape genetics. Journal of Applied Ecology 44:714-724.
17	Provide, if available, citations on Nelson's BHS (study cited on pg. 4-9 is for Sierra Nevada BHS)	Applicant	There are two citations on pg. 4-10, and several more in Section 5.2.2.5.
18	Calculate acreages for areas that will be mowed, non mowed, and allowed to re-vegetate	Applicant	Approximately 2200 acres (26%) of the site vegetation will remain intact and approximately 4400 acres will be mowed or have a local access road associated with solar arrays.
19	Provide the citation on the effects of shade from solar facilities	Applicant / CEC	See attached articles by S.D. Smith (1984 and 1987) provided as Attachment E.

Action Items from the Solar One January 5th, 2010 Workshop Continuation

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Item Number	Action Item	Responsibility	Response
20	Provide information on the ability of desert tortoises to use the northern end of the site with the debris basins as a movement corridor including the topography of the debris basins	Applicant	The basins themselves are not themselves the only movement corridor area. By keeping the basins unfenced, the basins provide extra space that can be used in addition to the area that is already available for movement. The basins will have three slopes of 4:1, while the slope facing the mountains (and corridor) will be 8:1. These slopes are gradual enough to allow passage by desert tortoise, as demonstrated by Attachment G, provided behind this response.
21	Provide pictures of the MFTL-occupied sand dune on the Solar One site	Applicant	Please see the photographs provided as Attachment F.
22	Provide information on the quantity, expected water quality, and duration of water presence in the evaporation ponds	Applicant	The current proposed water source is being chemically evaluated and the analysis is due the week of January 25, 2010. The volume and quality of the discharge from the reverse osmosis equipment is dependent upon the quality of the water being processed. We expect the water quality to be good enough to allow us to use 70% or better of the water being treated. Should the water quality prove to be good enough, the evaporation ponds will not be required, and the water will go directly to the de-mineralization unit.
23	Provide information on the feasibility of moving some of the plants found on-site. Prepare a draft relocation plan.	Applicant	Succulents and potentially crucifixion thorne may be able to be moved. Seed from some rare annuals may be collected prior to disturbance if site conditions during the spring are good. A draft relocation plan will be prepared by the Applicant and is expected to be submitted to the agencies for review during Mach of 2010.
24	Provide additional information on the Townsend's big ear bat sighting	Applicant	No specific information on the location/behavior of the observed Townsend's big ear bat is available.
25	Provide additional information on the abundance and distribution of weed species, and if available, provide a figure showing the information	Applicant	Abundance and specific locations of weed species present throughout the site was not noted.
26	Provide additional information on Table 4 of the Baseline Report in comparison with the USFWS numbers	Applicant	A detailed description of the method used to estimate the tortoise numbers presented in Table 4 of the Baseline report is presented in the fourth complete paragraph on pg. 4-5. The USFWS estimate is created by plugging information regarding the number of tortoise seen during surveys, the overall area surveyed and a probability that a tortoise is above ground variable based on the amount of rainfall from the winter previous of the survey. This number is 0.80 if the winter rainfall was ≥ 1.5 inches or 0.60 if the winter rain was < 1.5 inches. In 2007 the winter rains were 0.51inches, while the winter rain was 2.31 inches in 2008.
27	Provide clarification on the meaning of the "Alert" column in Table 1 of the weed management plan	Applicant	The alert column refers to a designation given to weeds from Cal-IPC. It is defined as "specific combinations of section scores that indicate significant potential for invading new ecosystems triggers an Alert designation so that land managers may watch for range expansions." from Cal-IPC website: http://www.cal-ipc.org/ip/inventory/index.php. Please see a revised Table 1 with explanation of Alert column, provided as Attachment B.

Action Items from the Solar One January 5th, 2010 Workshop Continuation

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Item Number	Action Item	Responsibility	Response
28	Provide copies of monitoring reports showing successful Burrowing Owl relocation following the consortium's guidelines if readily available	Applicant	URS has experience constructing and monitoring a successful Burrowing Owl relocation project, however, a monitoring report was not necessary or created for this project. Photos depicting use of constructed burrows by burrowing owls are provided as Attachment C.
29	Provide additional information on how the cell-by-cell surveys were temporally spaced (plants)	Applicant	2007 and 2008 rare plant surveys were conducted across the site in an east to west direction. The entire site was covered within a two week period each year.
30	Provide contact information for Gary Thomas	Applicant	Gary Thomas, Email: g.cranky@verizon.net, Phone: 951-206-6410
31	Perform surveys and submit reporting on Burrowing Owls on the Solar One site	Applicant	URS is currently conducting Burrowing Owl surveys, which will be completed by February 5, 2010. The Applicant will forward preliminary results once the fieldwork is finished and a formal report is expected to be available February 26, 2010.
32	Provide Streambed Alteration analysis as soon as it is available	Applicant	The Streambed Alteration Agreement application is anticipated to be submitted on February 19th, 2010
33	Provide documentation substantiating the applicant’s estimate of likelihood of finding rare plants during the 2007 and 2008	Applicant	The 2007 field effort resulted in no rare plant detections. This was due to low rainfall conditions, which is the reason why the 2008 survey was conducted. Rainfall was 88% of normal in 2008 and rare plants were detected in 2008. Years with above average rainfall would be expected to result in increased presence of rare plants known from the site and immediate project vicinity.
34	Provide more information about what is suitable habitat for rare plants in the Project study area.	Applicant	The 2008 distribution of detected rare plants are mostly associated with the alluvial fans and washes where flood flows occur during moderate to high rainfall years. Succulent species are not restricted to flood flow areas, and some of the rocky area can support certain cacti species.

Attachment A

Journal of Applied Ecology 2007
44, 714–724

Optimizing dispersal and corridor models using landscape genetics

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Summary

1. Better tools are needed to predict population connectivity in complex landscapes. 'Least-cost modelling' is one commonly employed approach in which dispersal costs are assigned to distinct habitat types and the least-costly dispersal paths among habitat patches are calculated using a geographical information system (GIS). Because adequate data on dispersal are usually lacking, dispersal costs are often assigned solely from expert opinion. Spatially explicit, high-resolution genetic data may be used to infer variation in animal movements. We employ such an approach to estimate habitat-specific migration rates and to develop least-cost connectivity models for desert bighorn sheep *Ovis canadensis nelsoni*.

2. Bighorn sheep dispersal is thought to be affected by distance and topography. We incorporated both factors into least-cost GIS models with different parameter values and estimated effective geographical distances among 26 populations. We assessed which model was correlated most strongly with gene flow estimates among those populations, while controlling for the effect of anthropogenic barriers. We used the best-fitting model to (i) determine whether migration rates are higher over sloped terrain than flat terrain; (ii) predict probable movement corridors; (iii) predict which populations are connected by migration; and (iv) investigate how anthropogenic barriers and translocated populations have affected landscape connectivity.

3. Migration models were correlated most strongly with migration when areas of at least 10% slope had 1/10th the cost of areas of lower slope; thus, gene flow occurred over longer distances when 'escape terrain' was available. Optimal parameter values were consistent across two measures of gene flow and three methods for defining population polygons.

4. Anthropogenic barriers disrupted numerous corridors predicted to be high-use dispersal routes, indicating priority areas for mitigation. However, population translocations have restored high-use dispersal routes in several other areas. Known intermountain movements of bighorn sheep were largely consistent with predicted corridors.

5. *Synthesis and applications.* Population genetic data provided sufficient resolution to infer how landscape features influenced the behaviour of dispersing desert bighorn sheep. Anthropogenic barriers that block high-use dispersal corridors should be mitigated, but population translocations may help maintain connectivity. We conclude that developing least-cost models from similar empirical data could significantly improve the utility of these tools.

Key-words: bighorn sheep, connectivity, fragmentation, GIS, isolation, least-cost model, metapopulation, *Ovis canadensis*, roads, translocation.

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Introduction

Defining and maintaining connectivity of natural populations has become a conservation priority (Moilanen *et al.* 2005). As natural populations become increasingly fragmented by habitat destruction and the creation of dispersal barriers such as roads, extinction probabilities for some populations will increase due to demographic and genetic factors associated with reduced dispersal (Hanski 1999; Hedrick 2005). Greater recognition that isolation of protected areas will lead to faunal relaxation (the gradual loss of species, e.g. Soule, Wilcox & Holtby 1979) has resulted in world-wide efforts to link protected areas using corridors, buffer zones and mixed-use areas. Models that incorporate land use, habitat quality, human activities and other factors are often employed to aid the mapping of landscape connectivity and prioritize land acquisitions (e.g. Hunter, Fisher & Crooks 2003; Nikolakaki 2004). However, identifying the optimal locations of wildlife corridors has proved to be difficult and controversial, in part because the details of how different species disperse across landscapes are often inadequately understood.

The advent of geographical information systems (GIS) analysis as a tool for identifying corridors and defining population connectivity has led to the widespread application of techniques such as 'least-cost' modelling (Adriaensen *et al.* 2003) and 'friction' analyses (Ray, Lehmann & Joly 2002; Joly, Morand & Cohas 2003; Sutcliffe *et al.* 2003; Nikolakaki 2004). Models created through these approaches are based typically on raster maps that divide landscapes into many cells with unique values that depict different habitat or vegetation types, elevation, slope or other landscape features. Cells are given weights or 'resistance values' reflecting the presumed influence of each variable on movement of the species in question. Least-cost routines (see Adriaensen *et al.* 2003), then, are employed to: (i) calculate the relative cost of all possible routes among populations or islands of core habitat; (ii) determine the least costly route for animal movement between pairs of populations or core areas of habitat; and (iii) plot these most probable routes on maps for use in conservation planning. 'Cost' is related to probability of transit and may not be defined explicitly; energetic costs, increased risk of predation or costs associated with reduced forage availability are among the reasons why an animal might avoid or be less able to traverse a landscape feature.

Although the least-cost approach has been employed widely (e.g. Adriaensen *et al.* 2003; Beazley *et al.* 2005; Rouget *et al.* 2006), this approach has two major drawbacks. First, the underlying models of dispersal (i.e. what resistance values are assigned to different landscape categories) are based rarely on anything more than informed opinions from experts. Where empirical data are available, dispersal costs are typically inferred from presence/absence or abundance data in different habitats, but such data may reflect habitat use rather than dispersal cost. Second, although these techniques define the most probable route according to the cost weighting system, the actual cost of a route over which dispersal can occur is unknown. Therefore, despite the increasing need and frequent application of such tools, these largely untested models are of uncertain value for conservation planning and management.

Population genetics approaches offer additional tools that can be applied to questions of dispersal and connectivity. Selectively neutral genetic markers can provide indices of gene flow derived from differences in allele frequencies between individuals or populations (Waser & Strobeck 1998). The emerging field of 'landscape genetics' uses high-resolution genetic data to determine the influence of landscape features such as fields (Vos *et al.* 2001) or highways (Keller & Largiader 2003; Epps *et al.* 2005) on gene flow and dispersal (Manel *et al.* 2003). However, developing dispersal models from genetic data entails large data sets and certain assumptions.

In particular, migration (in the sense of gene flow) operates at a different time scale than dispersal. Genetic data may reflect long-term dispersal patterns, but the time-period represented is variable and depends partly on the effective size (N_e) of the populations. Time to equilibrium between migration and drift is proportional to N_e (Slatkin 1993). Therefore, among populations with small N_e, estimates of genetic distance or gene flow should reflect more recent dispersal patterns than estimates among populations with large N_e. Simulated data can be used to describe more clearly the time scale for a given data set (e.g. Epps *et al.* 2005), but in general the time scale represented is unknown. Furthermore, migration reflects effective dispersal, i.e. dispersal followed by reproduction. Individuals that disperse and do not reproduce will not be represented unless they are sampled directly. This could be advantageous if effective dispersal is the process of interest, but might not be as useful when considering, for instance, the role of dispersing individuals in spreading disease. Finally, sex-biased dispersal must be considered; gene flow estimates derived from nuclear DNA may largely represent movements of the least philopatric sex. Despite these possible limitations, genetic analyses may provide comprehensive pictures of dispersal that are otherwise unavailable (Koenig, VanVuren & Hooge 1996).

Efforts to develop more sophisticated models of migration from genetic data that consider species' dispersal behaviour are increasingly common. One such approach is to examine the correlation of gene flow with measures of 'effective geographical distance' (EGD) among populations, in addition to measures of geographical distance or the presence or absence of specific elements such as roads (Michels *et al.* 2001). EGD is a composite measure of dispersal distance between populations that incorporates both geographical distance and landscape features hypothesized to affect dispersal. Recent examples of EGD include distances along riparian areas (Vignieri 2005), elevation change (Spear *et al*. 2005) and least-cost models that use a cost weighting surface based on assumed habitat value (Coulon *et al*. 2004; Spear *et al*. 2005; Vignieri 2005). EGD often explains more variation in gene flow between individuals or populations than geographical distance alone. This suggests that gene flow and dispersal patterns may not always fit a simple nearest-neighbour model, and it is important to test alternate hypotheses. However, genetic-based studies of dispersal rarely have examined more than a few alternate models of dispersal, and efforts to combine least-cost models with genetic data have been limited by a priori assumptions used to build the models. For instance, Vignieri (2005) used knowledge of preferred habitat for the Pacific jumping mouse *Zapus trinotatus* Rhoads to assign a lower dispersal cost to riparian and low-elevation habitat; however, that dispersal cost appeared arbitrary with respect to magnitude.

We propose that the effectiveness of combining least-cost and genetics-based approaches can be tested by comparing the ability of multiple least-cost models based on different landscape characteristics and a range of parameter values to explain observed variation in gene flow. Past analyses appear only to have tested hypotheses about which landscape factors affect dispersal. To translate least-cost models into effective conservation tools that identify active movement corridors and rank them according to predicted levels of gene flow, we also propose to estimate empirically how gene flow varies with EGD and determine the maximum EGD over which gene flow will occur.

In this paper we present methods to (1) test assumptions underlying least-cost connectivity models using genetic data; (2) predict landscape connectivity; and (3) test alternative management scenarios. We use estimates of gene flow among populations of desert bighorn sheep *Ovis canadensis nelsoni* Merriam to test the effectiveness of different least-cost GIS models and to optimize parameter values. We employ the following: (1) two methods for estimating gene flow among populations; (2) estimates of EGD derived from least-cost GIS models based on slope and distance with a wide range of parameter values; (3) three methods of defining population polygons used as the basis of our spatial analyses; (4) partial Mantel tests to assess correlation between gene flow estimates and EGD from alternate least-cost models; (5) regression of gene flow estimates on EGD to determine the maximum EGD over which gene flow is detectable; (6) identification and ranking of dispersal corridors using the best-fitting model of EGD; and (7) use of that model to identify probable movement corridors among populations of desert bighorn sheep while considering alternate management scenarios. Finally, we discuss the application of these techniques to conservation and management of species occupying fragmented habitats.

DESERT BIGHORN SHEEP AND PREVIOUS DISPERSAL MODELS

Desert bighorn sheep are desert-adapted ungulates native to the south-western United States. Preferred habitat is generally steep, rocky, arid terrain. In California, desert bighorn sheep populations are typically small, often < 50 individuals (Epps *et al*. 2003) and located in small mountain ranges isolated by varying expanses of low-lying desert habitat. The metapopulation-like distribution of desert bighorn sheep results in frequent extinction and recolonization of populations (Schwartz, Bleich & Holl 1986; Bleich, Wehausen & Holl 1990), and it is recognized that appropriate management requires consideration of population connectivity (e.g. determining when translocation of bighorn sheep may be needed to re-establish recently extirpated populations; Bleich *et al*. 1996). Bleich *et al*. (1996) proposed a model of population connectivity that considered populations < 15 km apart as likely to be connected by dispersal and hypothesized that interstate highways were barriers to dispersal. That model was used to determine management units above the level of individual populations. Low-resolution genetic markers [mitochondrial DNA (mtDNA) control region restriction fragment length polymorphism (RFLP) data] were used to verify that detectable genetic differences existed between management units.

Population genetics data from 26 populations of desert bighorn sheep in the Mojave and Sonoran Desert regions of California were used to investigate the spatial scale of gene flow and the role of anthropogenic (human-made) barriers such as interstate highways, urban areas and canals (Epps *et al*. 2005). Epps *et al*. (2005) tested whether estimates of gene flow and genetic distance (Nm and F_{ST}) were correlated with simple linear distance between populations and the presence of anthropogenic barriers. Those analyses confirmed that little or no gene flow had occurred across those barriers and that gene flow occurred primarily among populations < 15 km apart. However, habitat features expected to favour bighorn dispersal (e.g. areas with topographic relief sufficient to provide escape terrain for predator evasion) were not considered. Owing to considerable variation in the amount of escape terrain in low-lying areas among populations, we hypothesized that a least-cost model of migration based on topography could significantly improve our ability to predict the degree to which populations are linked by dispersal.

Materials and methods

OVERALL APPROACH: USING GENETIC DATA TO OPTIMIZE PARAMETER VALUES FOR A LEAST-COST MODEL

We used a matrix-based regression approach to test whether gene flow among populations of desert bighorn sheep varied as a function of distance and topography or distance alone, and to identify which model of distance and topography best approximated the effect of these variables on gene flow. First, we calculated a series of matrices ($\mathbf{X}_1$–$\mathbf{X}_i$) of effective geographical distances (EGD) among populations. Each matrix represented estimates of EGD between all population pairs among 26 populations of desert bighorn sheep in California, USA (Fig. 1), resulting from a unique set of parameter values (*i* unique combinations). Next, a matrix (**Y**) depicting the presence or absence of anthropogenic barriers (fenced highways, canals and urban areas) among those 26 populations was generated to control for the effect of those barriers on gene flow. Finally, a matrix (**Z**) of gene flow estimates between all population pairs was developed. We used partial Mantel tests to assess the correlation of **Z** (gene flow) with each matrix $\mathbf{X}_i$ (EGD), while controlling for the effect of **Y** (anthropogenic barriers). In that manner parameter values for the EGD model resulting in the strongest correlation between **X** and **Z** were identified. That exercise was repeated using three different methods to define the geographical extent of each population, as well as a second method of estimating gene flow, to examine how sensitive model fitting was to those variables. The optimized model of EGD was then used in later analyses of corridor length and location. Our methods are detailed in the following sections.

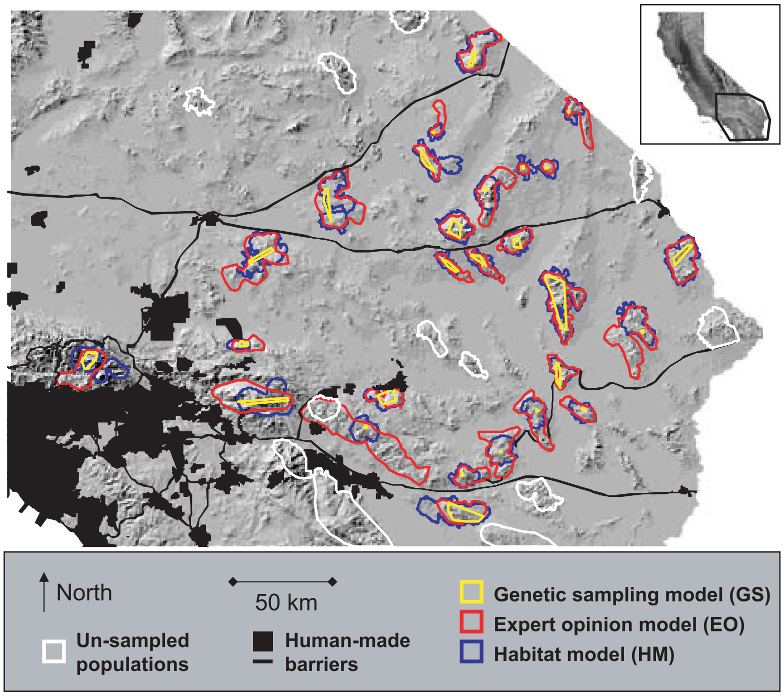

Fig. 1. Topography (hill-shade) and distribution of desert bighorn sheep in south-eastern California, United States. Coloured polygons represent genetically sampled populations used to develop the dispersal model. GS polygons are minimum convex polygons around genetic sample locations. EO polygons were hand-drawn based on topography and expert opinion on bighorn sheep distribution. HM polygons were developed either from a GIS habitat model (described in Appendix S2) or from 95% density kernels based on radio-telemetry locations. Population polygons not used for model development (outlined in white) are based on the HM or EO models. Anthropogenic barriers indicated include fenced interstate highways, canals and urban areas.

DEVELOPING LEAST-COST GIS MODELS TO CALCULATE EGD

We used slope as the variable for identifying the relative resistance or migration value of habitat between population polygons. We compiled 30 m Digital Elevation Model (DEM) data [US Geological Survey (USGS) 2003 series] for our study area and estimated slope for each 30 m cell using ArcGIS 9·0 (ESRI, Redlands, CA, USA). To simplify the models of bighorn migration as a function of topography and distance, we defined a 'slope cut-off' value for each model. Grid cells with slope greater than the cut-off value ('slope' cells) were considered more suitable (lower resistance) for bighorn dispersal than grid cells with slope lower than the cut-off ('flat' cells). We tested three slope cut-off values (5%, 10% and 15%), based on our assessment of radio telemetry data that suggested bighorn sheep are found mainly in habitat of at least 10% slope (3386 locations across the study area; unpublished data; California Department of Fish and Game). For each cut-off value tested, we generated six grids representing a wide range of different resistance values (weights) for slope cells. Thus, relative to the fixed cost of '1·0' for a flat cell, slope cells were given weights of 0·7, 0·5, 0·3, 0·1, 0·05 or 0·01 for each respective cost grid, yielding 18 different least-cost models and thus 18 matrices of different estimates of EGD ($\mathbf{X}_i$). For example, the model of EGD with 15% slope cut-off and slope cell weight of 0·1 considered cells with slope < 15% as 10 times more costly to cross than cells with slope > 15%. Slope grids were resampled at 90 m resolution to reduce calculation time.

ESTIMATING GENETIC DISTANCE AND GENE FLOW AMONG POPULATIONS

We used genetic data from 26 populations of desert bighorn sheep in California to develop the matrix of population pairwise gene flow estimates (**Z**). We identified 392 different individuals from data for 14 microsatellite loci using DNA extracted from faeces, tissue or blood, using two to six replicate polymerase chain reactions (PCRs) (see Epps *et al*. 2005). We used ARLEQUIN (Schneider, Roessli & Excoffier 2000) to estimate population pairwise F_{ST} values and transformed these to *Nm* values via the standard Wright–Fisher model $F_{ST} = 1/(1 + 4\,Nm)$ as our primary index of relative gene flow. Due to the restrictive assumptions of this model, *Nm* is unlikely to represent the actual number of migrants per generation (Whitlock & McCauley 1999) but can indicate relative levels of gene flow, particularly when migration rates exceed mutation rates (Slatkin 1993).

As a second measure of gene flow, we estimated migration rates (*M*) using MIGRATE (Beerli & Felsenstein 2001). Because computation time for the full data set of 26 populations was estimated at about 2 years, we restricted analyses to a subset of nine populations. MIGRATE estimates migration rates among populations using maximum-likelihood Markov chain Monte Carlo (MCMC) methods, and is an effort to improve migration rate estimates beyond the usual F_{ST}-based statistics (see Appendix S1 in Supplementary material for details).

USING GENE FLOW ESTIMATES TO TEST ALTERNATIVE PARAMETER VALUES

We used PATHMATRIX (Ray 2005) to calculate the least-cost paths among the 26 genetically sampled populations. This extension for ArcView version 3·2 (ESRI) uses a cost grid (here, derived from a given model of EGD) to (1) calculate least-cost paths among all pairs of population polygons; (2) generate the matrix $\mathbf{X}_i$ of EGD; and (3) map each least-cost path. Each estimate of EGD between a population pair is calculated as:

$$\text{EGD} = \sum x_j w_j \qquad \text{eqn 1}$$

where x_j is the linear distance across each grid cell j and w_j is the weight for that cell (determined here by whether the slope value is above or below the slope cut-off), summed over all the cells in a given path. All possible paths are evaluated, but only the EGD of the least-costly path is reported in matrix $\mathbf{X}_i$. Finally, we $\log_{10}$-transformed values in each matrix $\mathbf{X}_i$ to linearize the relationship of distance with *Nm* (Epps *et al.* 2005).

The presence of anthropogenic barriers (fenced highways, canals and urban areas) was found previously to affect gene flow strongly among these populations (Epps *et al.* 2005). We chose to correct for this effect by including barrier presence/absence as a second predictor matrix **Y** when assessing correlation between EGD and gene flow. Otherwise, if barriers were incorporated into each least-cost grid during the model-fitting process (by assigning large cost values to any grid cell where a barrier was present), appropriate cost values would vary for each least-cost grid. Inappropriate cost values would disrupt the otherwise linear relationship between gene flow (*Nm*) and $(\log_{10})$EGD. Moreover, those barriers have been present for only 40–60 years and have presumably affected gene flow at a different time scale than topography. Finally, barriers could be mitigated and therefore should be considered separately. We incorporated barriers formally into the underlying cost grid only when using the final best-fitting model to define active corridors (as described below). Barriers were recorded as present for any population pair with a barrier interposed; the map of barriers was compiled as described by Epps *et al.* (2005).

We used partial Mantel tests (Smouse, Long & Sokal 1986; Manly 1991) to estimate the partial correlation of matrix **Z** (*Nm* or MIGRATE *M*) with each matrix $\mathbf{X}_i$, while controlling for the presence of anthropogenic barriers by including matrix **Y** as a second predictor matrix. Tests were conducted using XLSTAT (Addinsoft, New York, USA). Partial Mantel tests determine the correlation of a response matrix **Z** to a predictor matrix **X**, while removing a spurious correlation resulting from a second predictor matrix **Y** that may be correlated with both **Z** and **X**. We used the value of the partial correlation coefficient r resulting for each $\mathbf{X}_i$ to compare graphically the relative fit of each model of EGD. We also estimated r for the null model $(\mathbf{X}_0)$ matrix of straight-line distances ($\log_{10}$-transformed) between population polygons.

While partial Mantel tests are controversial due to potential underestimation of type I error (Raufaste & Rousset 2001; Rousset 2002), Castellano & Balletto (2002) argued that this concern has been overstated. Moreover, because we compared the partial correlation coefficient of distance matrices while using the same second predictor matrix **Y** in all tests, and did not compare *P*-values, such underestimation is unlikely to affect our conclusions.

DEFINING POPULATION POLYGONS

Most metrics of gene flow use populations as the basic unit of comparison, defined theoretically as groups of freely interbreeding individuals. In practice, defining the spatial extent of populations may be difficult. To calculate accurate distances among populations, population map polygons must depict habitat used regularly by interacting individuals. To test how sensitive parameter optimization for the least-cost models was to population polygon definition, we repeated EGD calculations using three different methods to define population polygons.

Our first polygon model [‘Genetic sampling’ (GS); Fig. 1] used minimum convex polygons drawn around the locations in each mountain range where DNA samples were actually collected. If samples were collected at only one location such as a waterhole, we used a circle with diameter of 1 km centred on the sampling point. This approach would be useful for species where the extent of each population sampled is not defined clearly by the habitat patch and is likely to provide a conservatively small habitat area. The second polygon model [‘expert opinion’ (EO); Fig. 1] used the population polygons defined by Epps *et al.* (2005). These polygons were drawn on the basis of both the topographic extent of each mountain range and expert opinion regarding the distribution of bighorn sheep in each location, derived from field observations and helicopter surveys. Bleich *et al.* (1996) used a similar approach to define population polygons for management purposes. Expert opinion may often be the only available means to define populations for many species.

The final polygon model tested [‘habitat model’ (HO); Fig. 1] was a GIS model based on slope and distance to perennial water sources. It was designed to provide repeatable polygons depicting desert bighorn sheep distribution and to predict the probable distribution of new populations in vacant habitat. The model was developed using radio telemetry locations of desert bighorn sheep in five populations (California Department.

of Fish and Game, unpublished data) and expert opinion to inform model fit (see Appendix S2).

IDENTIFYING AND RANKING DISPERSAL CORRIDORS USING THE BEST-FITTING DISPERSAL MODEL

After examining graphically correlation coefficients from Mantel tests for all $\mathbf{X}_i$ matrices, repeated for three sets of population polygons and $\mathbf{Z}$ matrices based on two different estimates of gene flow, we chose the best-fitting model of EGD by selecting the values of slope cut-off and slope weight that resulted in the strongest correlation coefficients. We then used that best-fitting model to identify probable movement corridors between bighorn sheep populations, after selecting a population polygon model based both on performance and practical considerations.

To identify probable movement corridors, we used two regression-based procedures. We first estimated the maximum effective dispersal distance (the greatest effective geographical distance separating population polygons over which gene flow can be detected; hereafter, $\mathrm{EGD}_{\mathrm{MAX}}$) for desert bighorn sheep. This was performed via regression of population pairwise estimates of *Nm* on estimates of EGD from the best-fitting model for population pairs without intervening barriers. Gene flow, as measured by *Nm* between populations, is expected to decline with increasing distance until an asymptote at a 'background' non-zero level of *Nm* is reached. At distances greater than this point, current gene flow is unlikely but some degree of genetic similarity exists because of descent from common ancestors or recurrent mutations (Slatkin 1993). Because we could not identify a regression model that adequately described the rapid decline of *Nm* to a non-zero asymptote, we used XLSTAT version 2006.2 (Addinsoft) to perform nonparametric regression (Hardle 1992) of *Nm* on EGD from the best-fitting dispersal model. Nonparametric regression is essentially a smoothing method for predictive purposes. We used the LOWESS method with the tri-weight kernel and bandwidth equal to the standard deviation, based on the underlying model of a second-degree polynomial. We defined our estimate of $\mathrm{EGD}_{\mathrm{MAX}}$ as the point at which the predicted values from the nonparametric regression first stopped decreasing (excluding initial fluctuations at high *Nm*).

We defined active dispersal corridors as those least-cost paths with total cost $< \mathrm{EGD}_{\mathrm{MAX}}$. However, because nonparametric regression does not generate a general predictive equation for gene flow as a function of EGD, we modelled this relationship with a negative exponential regression function for $\mathrm{EGD} < \mathrm{EGD}_{\mathrm{MAX}}$ (where an adequate fit could be achieved) and used the resulting equation to predict relative gene flow over active dispersal corridors.

To identify probable dispersal corridors on the current landscape, we added barriers to the cost grid of the best-fitting migration model. Because Epps *et al.* (2005) determined that those barriers had eliminated recent gene flow, we assigned barrier cells a cost equivalent to $\mathrm{EGD}_{\mathrm{MAX}}$ to make them impermeable. After adding polygons for un-sampled populations to the population map, we used PATHMATRIX to calculate and map all least-cost paths between populations with a total cost less than $\mathrm{EGD}_{\mathrm{MAX}}$. This was repeated without human-made barriers in the cost-grid to examine how mitigation of those barriers might affect landscape connectivity. To investigate the role that translocations have played in maintaining population connectivity in south-eastern California, we repeated the first analysis but removed five populations re-established by the California Department. of Fish and Game through translocations. The relative strength of each corridor was assessed using the exponential decay model to estimate *Nm* as a function of EGD.

MODEL VALIDATION

Current radio-telemetry data were insufficient to validate the presence of dispersing bighorn sheep in the predicted least-cost corridor routes. Radio-telemetry locations were typically collected monthly; intermountain movements are relatively rare and time spent moving between mountain ranges may be of short duration. However, radio-collared or marked individuals have been detected after moving between mountain ranges. We compiled a list of all such movements as well as those inferred from anecdotal reports. We then evaluated whether least-cost paths from the best-fitting model linked each pair of ranges for which intermountain movements were detected.

Results

Effective geographical distance (EGD) based on topography was more strongly negatively correlated with gene flow (both *Nm*, as calculated from population pairwise F_{ST} values, and *M*, as estimated by MIGRATE) than straight-line distance in almost all cases, with an absolute increase of the correlation coefficient *r* of up to 23% (Fig. 2). EGD models based on 5% slope cut-off performed more poorly than models based on 10% or 15% slope in all cases. The 15% slope cut-off performed slightly better than the 10% cut-off over most (but not all) tests (Fig. 2). For all slope cut-off values, all population polygon models and both measures of gene flow, best-fitting models resulted when sloped terrain had 1/20th to 1/10th the cost of movement across flat terrain (Fig. 2), with the slope weight of 0·10 most often favoured. Therefore, the EGD model employing the 15% slope cut-off and slope weight of 0·10 (hereafter referred to as the 15/0·10 model) was used for further corridor modelling. Stronger correlation coefficients (*r*) were observed when using EO model population polygons (Fig. 2). However, the differences in *r* were not large, and

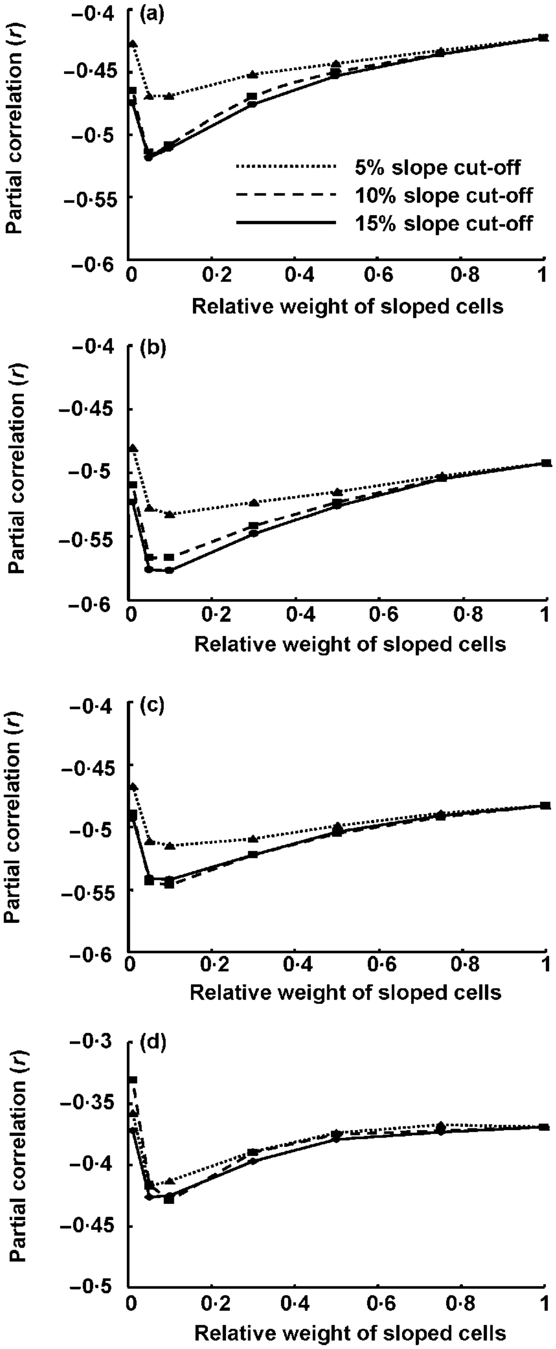

Fig. 2. Coefficients (r) for partial correlation of gene flow (Nm) with effective geographical distance from least-cost models, while correcting for anthropogenic barriers. Models use slope cut-off values of 5%, 10% and 15% and relative weights for slope cells of 0·01–1·0, for (a) GS polygons; (b) EO polygons; (c) HM polygons; and (d) a subset of nine populations using estimates of gene flow (M) from MIGRATE with HM polygons. The slope weight of 1·0 represents the shortest straight-line distance between population pairs.

optimal slope cut-off values and weights were similar, indicating low sensitivity to the choice of population polygon model. We chose HM polygons to calculate $\mathrm{EGD}_{\mathrm{MAX}}$ and model different corridor scenarios because this model can be used easily where bighorn sheep are currently absent or their distribution is poorly understood.

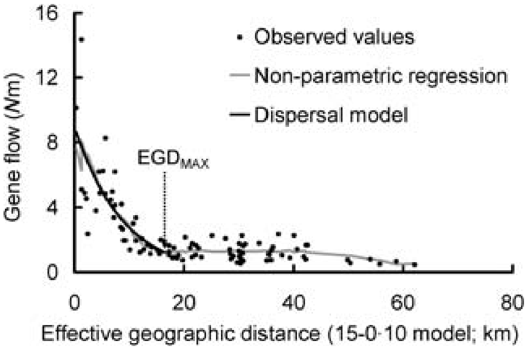

Fig. 3. Population pairwise estimates of gene flow (Nm) (for population pairs without intervening anthropogenic barriers) plotted against effective geographical distance (EGD) from the best-fitting model. Maximum effective dispersal distance ($\mathrm{EGD}_{\mathrm{MAX}}$, indicated with dashed arrow) was defined as the smallest EGD (after initial fluctuations) at which the slope of the line of predicted values generated by the nonparametric regression (grey line) stopped decreasing. Non-linear regression (black line) was conducted on all points below $\mathrm{EGD}_{\mathrm{MAX}}$ to generate a predictive model for gene flow as a function of EGD. Above $\mathrm{EGD}_{\mathrm{MAX}}$, dispersal was assumed to be negligible.

From nonparametric regression of population pairwise Nm values on estimates of EGD from the 15/0·10 model, we estimated the maximum effective dispersal distance ($\mathrm{EGD}_{\mathrm{MAX}}$) as 16·4 km-cost-units (corresponding to 16·4 km of flat terrain or 164 km of sloped terrain; Fig. 3). From regression of Nm values on EGD (km scale) for all values $< \mathrm{EGD}_{\mathrm{MAX}}$ (Fig. 3), we derived the following negative exponential model:

$$Nm = 9{\cdot}141 * e^{-0{\cdot}112 * \mathrm{EGD}} - 0{\cdot}219 \qquad \text{eqn 2}$$

We used equation 2 to estimate the relative strength of gene flow across active dispersal corridors with $\mathrm{EGD} < \mathrm{EGD}_{\mathrm{MAX}}$ (Fig. 4).

The connectivity of the current landscape suggested that nearly all populations are currently linked to another population by at least one possible dispersal corridor (black lines, Fig. 4a). However, in some cases these corridors had costs nearing $\mathrm{EGD}_{\mathrm{MAX}}$, making significant gene flow unlikely (narrow-width corridor lines, Fig. 4a). Comparison with corridors mapped in the absence of human-made barriers (yellow lines, Fig. 4a) indicated that those barriers have disrupted several regions of formerly high connectivity and resulted in complete isolation for at least one population. Mapping of corridors with and without populations re-established successfully by translocation (Fig. 4b) demonstrated that those translocations have helped maintain corridors for gene flow across a large region in the centre of the study area and several other areas, thereby greatly reducing the isolation of several native populations.

We identified 31 pairs of mountain ranges in the study area between which intermountain movements of bighorn sheep have been detected or inferred (Appendix S3). Of 22 pairs between which movements were detected via radio-telemetry or observation of marked animals, 21 pairs were linked by a predicted dispersal corridor. Of nine pairs between which movements were suggested on the basis of anecdotal reports, all were linked by predicted dispersal corridors.

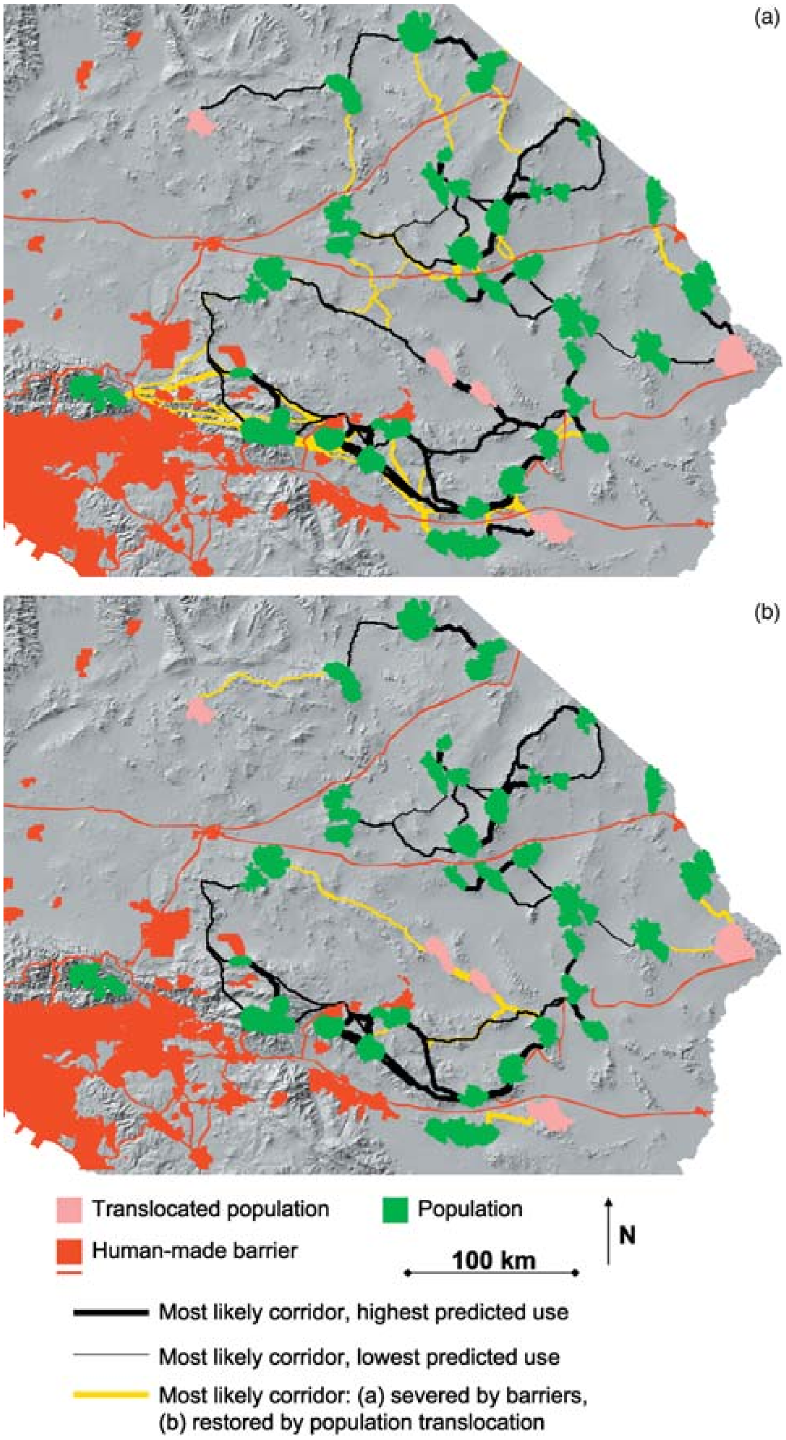

Fig. 4. Dispersal corridors predicted by the best-fitting dispersal model (15/0·10) and the HM population model, depicted with hill-shade topography. Black lines indicate least-costly corridor routes for corridors with cost < $\mathrm{EGD}_{\mathrm{MAX}}$, yellow lines indicate least-costly corridor routes that (a) were severed by anthropogenic barriers; or (b) were re-established by translocated populations. Corridors are presented based on (a) all extant populations within the study area, with and without current anthropogenic barriers considered; and (b) extant populations with and without those successfully re-established by translocation, with current anthropogenic barriers considered.

Discussion

Migration models that incorporated topography explained substantially more variation in gene flow than models that considered only geographical distance. While the models presented here reflect a small portion of possible models, we found that the best-fitting cost weights and slope cut-off values were consistent across different population polygon models and different measures of gene flow (Fig. 2). While time-consuming, we suggest that testing more than one type of gene flow estimate or population polygon model is important as a sensitivity analysis. Greater confidence in our results was derived from the concordance among models tested.

Inferring active dispersal corridors via the best-fitting migration model for desert bighorn sheep in California resulted in several conclusions. Most importantly, anthropogenic barriers currently fragment several regions that previously exhibited high connectivity (Fig. 4a), suggesting priority locations for the mitigation of these barriers. Additionally, mapping dispersal corridors including populations re-established by translocation (Fig. 4b) demonstrated that our models can be used to improve connectivity: if population establishment in an empty habitat patch could link existing populations by active dispersal corridors, a population translocation to that patch might receive higher priority. Potential future barriers can also be evaluated explicitly in this manner and avoided or mitigated at the time of construction. Finally, the successful restoration of several major dispersal corridors connecting otherwise isolated populations suggests that translocation could be used to restore critical nodes of population connectivity for other species.

These applications of the best-fitting migration model demonstrate the value of this tool for conservation and management. Because we parameterized this model from real data, we can have higher confidence that it models correctly the behaviour of bighorn sheep. We suggest reporting the effective geographical distance (EGD) values or predicted relative gene flow to rank corridors. Here, we scaled corridor widths by *Nm* to portray relative predicted corridor effectiveness (Fig. 4).

Comparison of the population polygon models suggested that, in this case at least, the definition of population extent did not affect greatly the parameterization of the migration model. Even the most restrictive polygon model (GS polygons, based on the location of the genetic samples collected) exhibited model-fitted curves of the same shape as those generated by the EO and HM polygons. This suggests that fitting least-cost dispersal models may be possible even in situations where the geographical extent of populations is difficult to define. If there is no clear basis at all for defining populations, it should also be possible to develop models in this fashion based on individual pairwise genetic comparisons (e.g. Vignieri 2005). Because this model-testing exercise was designed to examine migration, we caution against over-interpreting differences in absolute model fit between the population polygon models.

The number of populations in the genetic data set (26) was large, and such a sample might be considered prohibitive to applying this technique for other species. However, results obtained from testing dispersal models using MIGRATE *M* estimates for the nine-population subset were entirely consistent with those from the full data set (Fig. 3d). Thus, even relatively few populations may suffice to fit such dispersal models.

The connectivity network derived from the genetic analyses confirmed that knowledge of bighorn sheep behaviour (i.e. preference for steep terrain) could be incorporated into a connectivity design, even to the extent of identifying where additional population nodes could be reintroduced to improve the overall connectivity of the bighorn sheep metapopulations. This, in turn, suggests that core and corridor analyses for other species, based on behaviour and proper weighting of landscape variables, could provide important tools for management and conservation. Many aspects of this approach bear further investigation. For instance, rather than use the cumbersome 'trial and error' testing of model parameters, it may be possible to determine the best-fitting model mathematically. However, no mathematical solution will be possible once the number of parameter estimates exceeds the number of population pairs with genetic data. Setting up a few biologically plausible alternative models for testing and exploring restricted subsets of parameter space may be the most practical strategy.

Another aspect worthy of investigation is how best to determine when one model represents a 'significant' improvement over another. Model-selection techniques such as Akaike's information criterion (AIC) may be of little value when the identity of the predictor variables does not change among models. For this reason, we selected the best models using a graphical assessment of model fit. In the end, once the appropriate range of model parameters is identified, slight variations in model fit resulting from small differences in cost weights are likely to be unimportant. In our case, fitting corridors based on slope supported the behavioural inference that bighorn sheep prefer to travel over sloped terrain offering security from predators, regardless of minor differences between 10% and 15% slope cut-offs. Small changes in model parameters may become more important when considering whether an individual corridor is likely to be used or not. For this reason, we reiterate that the relative likelihood of corridor use should be considered, rather than merely a 'corridor or not-corridor' assessment.

MODEL VALIDATION

Known intermountain movements by bighorn sheep correlated well with our corridor model, with the exception of one marked individual that apparently crossed an interstate highway. This observation highlights the difference between individual dispersal events and the broad patterns of movement over time inferred by our analyses of gene flow. Occasional movements may far exceed those predicted by our migration model. Whether bighorn sheep follow routes consistent with the least costly paths among ranges is also unclear. Acquiring enough data points to verify the complete movement paths of dispersing bighorn sheep will probably require the use of GPS collars set to collect multiple locations per day. Until then, path locations predicted by our model should be considered as hypotheses for further testing.

LIMITATIONS OF THE APPROACH

While the field of landscape genetics is making rapid strides in developing analyses of gene flow that consider complicated landscape features, our approach still has a number of limitations. For instance, such a modelling exercise is better suited to dealing with common landscape characteristics that affect large numbers of populations, given the low statistical power of matrix correlation tests. In this analysis, the south-westernmost populations inhabit mountain ranges with thick forests and chaparral. Those habitat elements probably strongly limit movement by bighorn sheep because of increased predation risk. We did not consider those elements in model development because of the small number of populations affected; thus, connectivity in that region may have been overstated.

A second limitation to our model is that it reflects more effectively the potential for gene flow rather than colonization of empty habitat patches. Desert bighorn sheep have sex-biased dispersal: males are much more likely to travel long distances between populations, while females are probably the limiting factor in colonization events. Because the model described here is fitted using nuclear genetic markers, it represents both male- and female-mediated gene flow. A correction for the reduced movement of females possibly could be generated from radio-telemetry data or mtDNA, although the variability in estimates of gene flow from mtDNA (resulting from its behaviour as one linked locus) makes its use inherently imprecise. This limitation may be important to consider when using these models for management decisions; for example, determining when translocation may be necessary for population re-establishment.

Determining how to model landscape features such as anthropogenic barriers proved to be a complex issue. We dealt with those barriers in a separate analytical framework during model fitting and brought them back into the final model. This approach seemed appropriate because roads have been present on the landscape for only a short period of time. Moreover, road impacts can be mitigated and therefore corridor design should be assessed as a function of the mitigated landscape. A further technical limitation is that the width of interstate highway corridors and other barriers varies; ideally, the estimated cost of the barrier should be applied to any path crossing the barrier but not on a per-pixel basis (where that cost is accumulated for each pixel encountered). Other, more integrative approaches may be of value in other systems.

Finally, an important caveat is that we used migration, a long-term process, to make inferences about current patterns of bighorn sheep dispersal. Variation in allele frequencies used to estimate migration may be affected by other factors such as population bottlenecks (Whitlock & McCauley 1999). Moreover, if past conditions are reflected more strongly than current dispersal patterns, management decisions using these models might be flawed. However, the small size of these populations and the detectable effect of barriers present for only six to seven generations (Epps *et al.* 2005) suggest that in this case we can still make useful inferences about movement of bighorn sheep on the recent landscape as well as identify factors likely to affect connectivity on the current landscape. Because dispersal is a complex process and the reasons that an individual animal does or does not disperse are unclear, and may not be reduced to simple models, fitting least-cost models using genetic data is probably most effective at identifying broad-scale patterns of gene flow resulting from landscape features that have been present for at least a few generations.

IMPROVING CORRIDOR MODELS AND PLANS TO MAINTAIN OR RE-ESTABLISH CONNECTIVITY

Our study suggests that developing least-cost models from genetic data can improve significantly the quality of and confidence in models of dispersal, migration and connectivity. Other types of data on movement could be used in a similar approach (e.g. Sutcliffe *et al.* 2003). Least-cost models have been employed worldwide to plan landscape-scale conservation strategies, to design reserves and to assess the effects of habitat fragmentation on many species. In some cases those models may have been applied uncritically with respect to their underlying assumptions. While developing genetic data or other data on movement may be a difficult task for many species, it may at least be possible to inform such models using data from species with similar biological characteristics.

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Supplementary material

The following supplementary material is available for this article.

Appendix S1. Estimating migration rates *M* using MIGRATE software.

Appendix S2. Defining the Habitat Model population polygons.

Appendix S3. Reported intermountain movements by desert bighorn sheep in the study area.

This material is available as part of the online article from: http://www.blackwell-synergy.com/doi/full/10.1111/j.1365-2664.2007.01325.x

Supplementary Appendix 1: Estimating migration rates *M* using MIGRATE software

Each of three repeated runs used the microsatellite (ladder) mutation model with the default search strategy (ten short and three long chains, burn-in of 10,000). MIGRATE (Beerli & Felsenstein 2001) reports migration rates (*M*, which is the migration rate *m* divided by an estimated constant mutation rate) both "from" and "to" all populations. For each population pair in the nine-population subset, we averaged both estimates to obtain a single migration value for each run. Migration values across the three runs were then averaged for the final estimates of pair-wise population migration rates.

References

Beerli, P. & Felsenstein, J. (2001) Maximum likelihood estimation of a migration matrix and effective population sizes in *n* subpopulations by using a coalescent approach. *Proceedings of the National Academy of Sciences of the United States of America*, **98**, 4563-4568.

Supplementary Appendix 2: Defining the Habitat Model population polygons based on cost distance with respect to slope and perennial water sources

While not intended to predict fine-scale habitat use, this model was used to define the general areas likely to experience significant use by both sexes of this species. Although core habitat areas are usually defined by using a land cover map to identify preferred habitat (e.g. Thorne *et al.* 2006), desert bighorn sheep use steep terrain almost exclusively. Therefore, we identified habitat on the basis of slope and distance to perennial water sources rather than vegetation type.

We defined habitat as all areas within 8,000 cost units (in meters) of perennial water sources available to bighorn sheep. We used a cost grid where flat terrain (<10% slope) had a relative cost of "15", sloped terrain (10% slope or greater) had a relative cost of "1", and human-made barriers (interstate and other fenced sections of divided highways, canals, and urban areas) were assigned a relative cost of "10,000" to ensure that no areas separated from perennial water by such barriers were included as habitat. A map of perennial water sources used by desert bighorn sheep was developed to serve as the basis for this habitat model, using GPS coordinates and field notes compiled by the California Department of Fish and Game (CDFG), the first author, and other knowledgeable individuals. We used the ArcGIS 9.0 "*Cost Distance*" model to identify all cells meeting the above definition of habitat, and then used the following operations to generate simplified polygons based on those "habitat" cells (ArcGIS 9.0 operations are specified in italics):

1) *Majority Filter* [redefines pixels on the basis of the identity of at least 2 of the 4 pixels bounding the sides of the pixel in question, with preference given to "habitat" pixels in the event of a tie; smoothes and simplifies polygons]
2) *Boundary Clean* [expands and contracts the boundary of the habitat pixels twice, results in further smoothing]
3) *Raster to Polygon* [to allow any needed adjustments to polygon lines in step 4]
4) Manual adjustments in one case to merge two polygons that were only ~100 m apart, thereby creating one polygon for the mountain range in question
5) *Polygon to Raster*
6) *Raster to Line Feature* [to enable deletion of "bubbles" in step 7]
7) Manual deletion of "bubbles" (tiny areas of "non-habitat" enclosed in larger "habitat" polygons)
8) *Feature to Polygon*

We examined two slope cut-off values for habitat (10 and 15%) and tested several relative slope weights (data not shown), but chose the above values of slope, slope weight, and maximum cost distance as qualitatively most representative of desert bighorn sheep distribution (as inferred from radiotelemetry locations in five desert populations and field observations).

We considered the habitat model to be applicable only for populations in desert mountain ranges lacking dense forested vegetation. Three populations (the San Gabriel, San Gorgonio, and Cushenbury populations in the southwestern corner of the study area, Fig. 1 of main manuscript) occurred in mountain ranges with significant portions of habitat covered in dense forests or chaparral and, therefore, were less suitable for bighorn sheep. We defined population polygons for those three populations based on radiotelemetry locations (CDFG, unpublished data; monthly locations are derived from males and females totaling as follows: 28 individuals over 3 years (San Gabriel), 7 individuals over 1 year (San Gorgonio), and 25 individuals over 8 years (Cushenbury)). We used a fixed kernel density estimator (Beyer 2004) to define the 95% density kernel, and increased the amount of smoothing until a single 95% density polygon was achieved for each of those three populations. Those polygons were added to the population polygons generated by the habitat model to create the final "Habitat Model" polygons used in dispersal model analyses.

References

Beyer, H.L. (2004) Hawth's Analysis Tools for ArcGIS. Accessed April 21, 2006, http://www.spatialecology.com/htools.

Thorne, J.H., Cameron, D., & Quinn, J.F. (2006) A conservation design for the central coast of California and the evaluation of mountain lion as an umbrella species. *Natural Areas Journal*, **26**, 137-148.

Supplementary Appendix 3: Reported inter-mountain movements by desert bighorn sheep in the study area. Telemetry data were compiled by California Department of Fish and Game or National Park Service (Queen Mountain data). If evidence for inter-mountain movements was detected during review of radio-telemetry locations for this study (unpublished data, California Department of Fish and Game), but is not published elsewhere, source is listed only as "Radio-telemetry". Names listed without citations reflect personal communications.

Pairs of mountain ranges with recorded movements	Predicted by corridor model?	Type of observation and source
Kingston, Clark	Yes	Radio-telemetry
Old Dad Peak, Club Peak	Yes	Radio-telemetry
North Bristol, Cady	Yes	Radio-telemetry
North Bristol, Granite	Yes	Radio-telemetry
Granite, Old Dad Mtns	Yes	Radio-telemetry
Marble, South Bristol	Yes	Radio-telemetry
Marble, Clipper	Yes	Radio-telemetry (Bleich *et al.* 1990)
Sheephole, Coxcomb	Yes	Radio-telemetry (Bleich *et al.* 1990)
Eagle, Coxcomb	Yes	Radio-telemetry (D. Devine; also see Devine 1998)
Sheephole, Bullion	Yes	Radio-telemetry (Bleich *et al.* 1990)
Sheephole, Calumet	Yes	Radio-telemetry (Bleich *et al.* 1990)
Turtle, Whipple	Yes	Radio-telemetry (Bleich *et al.* 1990)
Old Woman, Turtle	Yes	Recovered carcass of marked animal (Bleich *et al.* 1990)
Old Woman, Iron	Yes	Radio-telemetry (Bleich *et al.* 1990)
Orocopia, Chuckwalla	Yes	Radio-telemetry
Lost Palm, Buzzard Spring (Eagle Mtns)	Yes	(Devine & Douglas 1996)
Old Woman, Ship	Yes	Radio-telemetry (Bleich *et al.* 1990)
Orocopia, West Chocolate	Yes	Radio-telemetry
Queen, Pinto	Yes	Radio-telemetry (K. Longshore)
Old Dad Peak, Cowhole	Yes	Radio-telemetry (Bleich *et al.* 1990)

Supplementary Appendix 3 (continued)

Old Dad Peak, Soda Mtns (south of Interstate 15)	Yes	Radio-telemetry (V. Bleich)
Clipper, Marl	No (separated by Interstate 40)	Observation of marked animal during helicopter survey (J. Wehausen)
Chemhuevi, Whipple	Yes	Anecdotal reports of sheep crossing intervening roads (Bleich *et al.* 1990)
Ivanpah Range, New York	Yes	Photos of sheep crossing intervening roads (V. Bleich)
Queen, Bullion	Yes	Anecdotal reports of sheep crossing intervening roads (Bleich *et al.* 1990)
Little San Bernardino, San Gorgonio	Yes	Anecdotal reports of sheep crossing intervening roads (Bleich *et al.* 1990)
Avawatz, North Soda	Yes	Anecdotal observations of sheep in Soda Mtns (V. Bleich)
Old Woman, Little Piute	Yes	Anecdotal reports (Bleich et al., 1990)
Granite, Providence	Yes	Anecdotal reports of sheep crossing intervening road (J. Andre)
Chemhuevi, Sacramento	Yes	Anecdotal reports of sheep crossing intervening road (D. Weaver)
Little San Bernardino, Eagle	Yes	Anecdotal reports of sheep crossing intervening road (J. Ashdown)

References

Bleich, V.C., Wehausen, J.D., & Holl, S.A. (1990) Desert-dwelling mountain sheep: conservation implications of a naturally fragmented distribution. *Conservation Biology*, **4**, 383-390.

Divine, D.D., & Douglas, C.L. (1996) Bighorn sheep monitoring program for the Eagle Mountain landfill project: phase I report. Unpublished report, submitted to California Department of Fish and Game.

Divine, D.D. (1998) Habitat patch dynamics of desert bighorn sheep *Ovis canadensis nelsoni* in the eastern Mojave Desert. Ph.D. Thesis, University of Nevada-Las Vegas.

Table 1
Observed and Potentially Occurring Noxious Weeds at Solar One Project Site

Scientific Name	Common Name	Habitats of Concern and Comments	Observed During Surveys and Anticipated Distribution in Project Area	CDFA Rank	Cal-IPC Overall Rating	Alert	Cal-IPC Impacts Rating	Cal-IPC Invasive Rating	Cal-IPC Distribution Rating
Ailanthus altissima	tree of heaven	Riparian areas, grasslands, oak woodland. Impacts highest in riparian areas.	Not observed; does not appear to propagate beyond old home and settlement sites.	Not Listed	Moderate	No	B	B	B
Alhagi camelorum	camel thorn	Grassland, meadows, riparian and desert scrub, Sonoran thorn woodland. Very invasive in southwestern states. Limited distribution in California	Not observed; monitors should attempt to remove this species if it is encountered in the future.	A	Moderate	No	B	B	B
Avena barbata; A. fatua	slender wild oats; wild oats	Coastal scrub, grasslands, oak woodland, forest. Very widespread, but impacts more severe in desert regions.	Not observed; monitors should attempt to remove this species if it is encountered in the future.	Not Listed	Moderate	No	B	B	A
Brassica tournefortii	Sahara mustard	Desert dunes, desert and coastal scrub	Observed; widespread and abundant throughout site. Control efforts should be made.	Not Listed	High	No	A	A	B
Bromus diandrus	Ripgut Brome	Dunes, scrub, grassland, woodland, forest. Very widespread, but monotypic stands uncommon.	Not observed; common non-native grass in California. High potential for this species to occur onsite in the future.	Not Listed	Moderate	No	B	B	A

Scientific Name	Common Name	Habitats of Concern and Comments	Observed During Surveys and Anticipated Distribution in Project Area	CDFA Rank	Cal-IPC Overall Rating	Alert	Cal-IPC Impacts Rating	Cal-IPC Invasive Rating	Cal-IPC Distribution Rating
Bromus madritensis ssp. *rubens*	red brome	Scrub, grassland, desert washes, woodlands	Observed; widespread but with patchy distribution throughout the Project area, often at the base of shrubs; Too extensive for control.	Not Listed	High	No	A	B	A
Bromus tectorum	downy brome, cheatgrass	Interior scrub, woodlands, grasslands, pinon/Joshua tree woodland, chaparral.	Not observed; high potential for this species to occur on the site in the future.	Not Listed	High	No	A	B	A
Cynodon dactylon	Bermuda grass	Riparian scrub in southern California. Common landscape weed, but can be very invasive in desert washes.	Not observed.	C	Moderate	No	B	B	B
Descurainia sophia	flixweed, tansy mustard	Scrub, grassland, woodland. Impacts appear to be minor, but locally more invasive in northeast California.	Not observed.	Not Listed	Limited	No	C	B	B
Elaeagnus angustifolia	Russian olive	Interior riparian. Impacts more severe in other western states. Current distribution limited in California.	Not observed; escapees widespread in Clark and Nye Counties, Nevada, but restricted to townsites and occasional riparian habitats.	Not Listed	Moderate	No	B	A	B
Erodium cicutarium	red-stemmed filaree	Many habitats. Widespread. Impacts minor in wildlands. High-density populations transient.	Observed; widespread and abundant onsite. Too extensive to be controlled.	Not Listed	Limited	No	C	C	A

Scientific Name	Common Name	Habitats of Concern and Comments	Observed During Surveys and Anticipated Distribution in Project Area	CDFA Rank	Cal-IPC Overall Rating	Alert	Cal-IPC Impacts Rating	Cal-IPC Invasive Rating	Cal-IPC Distribution Rating
Halogeton glomeratus	halogeton	Scrub, grasslands, pinyon-juniper woodland. Larger problem in Nevada. Monotypic stands are rare.	Not observed.	A	Moderate	No	B	A	B
Lactuca serriola	Prickly lettuce	Primarily an agricultural and roadside weed	Not observed.	Not Listed	Eval No List	No	D	B	C
Malva parviflora	cheeseweed	Common in disturbed places throughout California. More widespread in desert regions	Not observed.	Not Listed	Not Listed	N/A	N/A	N/A	N/A
Mesembryanthemum crystallinum	crystalline iceplant	Coastal bluffs, dunes, scrubs, grasslands. Limited distribution. Locally problematic, especially in southern CA.	Not observed.	B	Moderate	Yes	B	B	C
Phalaris minor	Mediterranean canary grass	Common in disturbed areas especially near washes. Widespread in low elevation California deserts.	Not observed.	Not Listed	Not Listed	N/A	N/A	N/A	N/A
Salsola paulsenii	barbed-wire Russian thistle	Desert and Great Basin scrub. Limited distribution. Impacts in desert appear to be minor.	Not observed; widespread but typically uncommon except in recently disturbed habitats.	C	Limited	No	C	C	C

Scientific Name	Common Name	Habitats of Concern and Comments	Observed During Surveys and Anticipated Distribution in Project Area	CDFA Rank	Cal-IPC Overall Rating	Alert	Cal-IPC Impacts Rating	Cal-IPC Invasive Rating	Cal-IPC Distribution Rating
Salsola tragus; S. kali; S. pestifer	Russian thistle; tumble weed	Desert dunes and scrub, alkali playa. Widespread. Impacts minor in wildlands.	Observed; widespread but with a patchy distribution.	C	Limited	No	C	B	B
Schismus arabicus, Schismus barbatus	Mediterranean-grass	Scrub, thorn woodland. Widespread in deserts. Impacts can be more important locally.	Observed; patchy distribution throughout the Project area.	Not Listed	Limited	No	B	C	A
Sisymbrium irio	London rocket	Scrub, grasslands. Widespread. Primarily in disturbed sites. Impacts vary locally.	Observed; patchy distribution throughout the Project site.	Not Listed	Moderate	No	B	B	A
Solanum elaeagnifolium	white horsenettle	Primarily agricultural weed, but escaping to wildlands in other countries. May be expanding range.	Not observed; can be expected to occur in Project area.	B	Eval No List	No	D	B	C
Sonchus oleraceus	Common sow thistle	Primarily an agricultural weed	Not observed.	Not Listed	Eval No List	No	D	B	B
Tamarix ramosissima; Tamarix sp.	saltcedar	Desert washes, riparian areas, seeps and springs	Observed onsite. Populations should be eradicated. Usually confined riparian areas, or other areas with available surface and/or ground water.	B	High	No	A	A	A

Scientific Name	Common Name	Habitats of Concern and Comments	Observed During Surveys and Anticipated Distribution in Project Area	CDFA Rank	Cal-IPC Overall Rating	Alert	Cal-IPC Impacts Rating	Cal-IPC Invasive Rating	Cal-IPC Distribution Rating
Tribulus terrestris	puncture vine		Not observed; it is likely that the next good summer rainy season should raise both *Solanum* and *Tribulus.*	C	Not Listed	N/A	N/A	N/A	N/A

CDFA Ratings:

“A” Eradication, containment, rejection, or other holding action at the state-county level. Quarantine interceptions to be rejected or treated at any point in the state.

“B” Eradication, containment, control or other holding action at the discretion of the commissioner.

"C" – State endorsed holding action and eradication only when found in a nursery; action to retard spread outside of nurseries at the discretion of the commissioner; reject only when found in a cropseed for planting or at the discretion of the commissioner.

"Q" – Temporary "A" action outside of nurseries at the state-county level pending determination of a permanent rating. Species on List 2, "Federal Noxious Weed Regulation" are given an automatic "Q" rating when evaluated in California.

Cal-IPC ratings:

High – These species have severe ecological impacts on physical processes, plant and animal communities, and vegetation structure. Their reproductive biology and other attributes are conducive to moderate to high rates of dispersal and establishment. Most are widely distributed.

Moderate – These species have substantial and apparent – but generally not severe – ecological impacts on physical processes, plant and animal communities, and vegetation structure. Their reproductive biology and other attributes are conducive to moderate to high rates of dispersal, although establishment is generally dependent on ecological disturbance. Ecological amplitude and distribution may range from limited to widespread.

Limited – These species are invasive but their ecological impacts are minor on a statewide level or there was not enough information to justify a higher score. Their reproductive biology and other attributes result in low to moderate rates of invasiveness. Ecological amplitude and distribution are generally limited, but these species may be locally persistent and problematic.

Alert – An “alert” is triggered when specific combinations of section scores indicate significant potential of the weed species for invading new ecosystems. This alert designation was created so that land managers will be aware to look out for range expansions these species.

Sources:
CDFA, 2004
Cal-IPC, 2006
Mojave Resource Conservation District, 2003

Attachment C

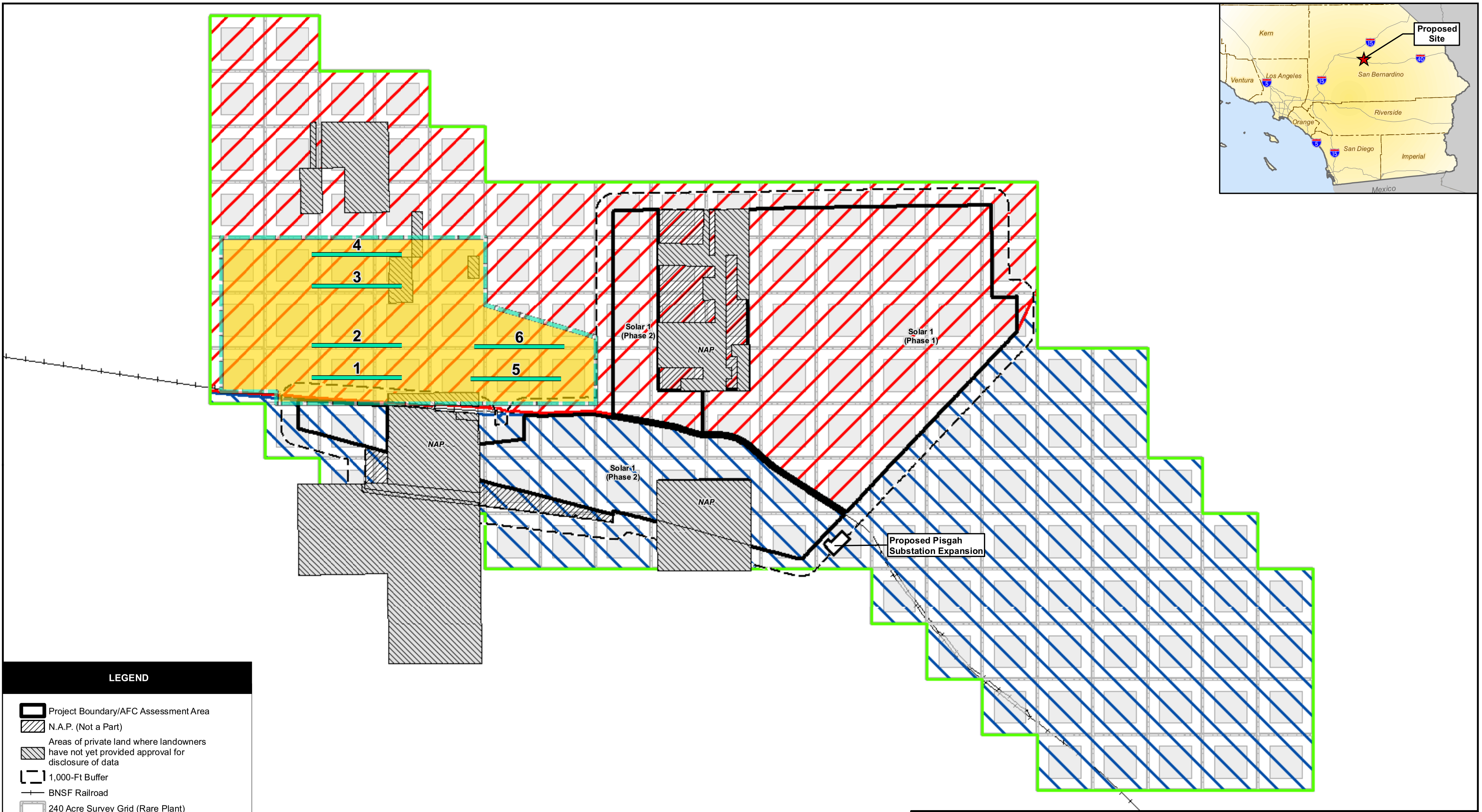
Photograph #1

Photograph #2

	**Photograph** #3
	**Photograph** #4

	**Photograph** #5
	**Photograph** #1

Attachment D



LEGEND

▭	Project Boundary/AFC Assessment Area
▨	N.A.P. (Not a Part)
▨	Areas of private land where landowners have not yet provided approval for disclosure of data
⌐ ¬	1,000-Ft Buffer
┼	BNSF Railroad
▭	240 Acre Survey Grid (Rare Plant)
▭	80 Acre Survey Grid (Desert Tortoise)
🟥	Surveyed In 2007 (Rare Plant/Desert Tortoise)
🟩	Surveyed In 2008 (Rare Plant)
🟦	Surveyed In 2008 (Desert Tortise)
▬	2009 Rare Plant Survey Area Transects
🟨	2009 Rare Plant Survey Area

SOURCES:
Huitt-Zollars, Inc (site plan Aug. 2009);
TIGER (railroad 2000); ESRI (overview);
URS (MFTL and sensitive species surveys).

2700 0 2700 5400 Feet

SCALE: 1" = 5400' (1:64,800)
SCALE CORRECT WHEN PRINTED AT 11X17

URS

URS SURVEY AREAS
SOLAR ONE PROJECT

CREATED BY: PM	DATE: 1-20-10	FIG. NO:
PM: WM	PROJ. NO: 27658189.20001	4

Path: G:\gis\projects\1577\27658100\mxd\Bio\bio_S1_urs_surveyarea_20100120.mxd, 01/20/10, paul_moreno

Attachment E

UCLA--12/1464

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ENVIRONMENTAL EFFECTS OF SOLAR THERMAL POWER SYSTEMS

ANALYSIS OF PLANT INVASION INTO THE BARSTOW 10 MWe PILOT STPS

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Abstract

An analysis of the invading vegetation within the heliostat field of Solar One was conducted during 1983 and 1984. Vegetation analyses included both floristic inventories and quantification of plant density, diversity, and aboveground biomass. An attempt was also made to assess present and potential operational/safety problems associated with vegetation presence on the site.

The invading vegetation within the heliostat field was both floristically and structurally dissimilar from an adjacent open desert control site. The invading flora was composed of primarily introduced, weedy annuals, whereas the open desert was characterized by native perennial forbs. Quantitatively, several trends emerged: (1) green biomass of ephemerals and newly germinated woody plants was highest in the control site in spring but in the heliostat field in the summer, (2) species diversity was consistently highest in the heliostat field, (3) average plant size was greater in the heliostat field, and (4) plant development and phenology was temporally shifted in the heliostat field, resulting in delayed senescence of plants into the dry season relative to the control site. Heliostat stow position and washing appear to influence the spatial pattern of vegetation in the heliostat field. The direct effects of shading and perhaps water addition appear to influence vegetation presence and structure to a greater degree than does previous clearing and surface disturbances.

The presence of vegetation apparently has caused operational and/or safety problems on the site, as the heliostat field was manually cleared or treated with herbicide during each year of the study. The primary problem appears to be tumbleweed, Salsola iberica, which grew abundantly in the heliostat field in the summer of 1983. This dense cover of Salsola, which was greatest near the base of each heliostat, prevented access to control boxes and provided possible cover for poisonous snakes.

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1. Introduction

Solar energy is one of the most promising solutions for the predicted energy shortages of the rapidly developing southwestern United States. This area has extensive periods of sunshine and little precipitation, which are prime attributes for development of large scale solar conversion facilities. As a result, plans are presently in effect to construct several large solar conversion facilities in the region, and many are anticipated for arid and semiarid regions of the world over the next several decades. The Department of Energy (DOE) and several regional utilities have constructed a pilot 10 MWe solar thermal power system (STPS) near Barstow, in San Bernardino County, California. This project, termed Solar One, is the first large central receiver facility in the United States.

Before solar energy conversion systems are developed on a large scale basis, it is important that the ecological changes created by these solar power plants be understood, at least qualitatively, prior to full scale implementation. Construction and operation of large solar conversion facilities will create a form of perturbation which is new to arid regions, and methods for amelioration and/or stabilization of these changes need to be established. It should be emphasized, however, that the ecological changes caused by solar conversion systems may not necessarily be deleterious but may be used in an economically beneficial fashion. This potential contrasts solar energy technology with other technologies now employed to generate electrical and chemical energy in the southwestern United States and other arid regions of the world.

The potential environmental effects of large scale solar thermal central receiver power systems have been reviewed recently (Davison and Grether, 1977; Lindberg, 1979; Turner, 1980). These studies conclude that land and water requirements of central receiver systems are the most important environmental issues associated with development of the resource in arid regions; mesoclimatic alterations are also considered to be potentially significant. Large scale development of photovoltaic solar power systems are expected to have similar land use and climatological consequences as the central receiver systems, but the photovoltaic systems may have greater materials-related environmental problems (Holdren et al., 1980).

Possible influences of solar thermal power plants on climatic parameters have been addressed in numerous hypothetical analyses, with the general conclusion being that these effects would be minor or nonexistent in areas adjacent to the facility (e.g., Turner, 1982). In contrast, micrometeorological parameters at or near the ground surface within heliostat fields may be substantially altered due to shading and wind deflection by the array of structures (Patten, 1978). Specifically, the results of a study utilizing an array of simulated heliostats in the Sonoran Desert of Arizona showed that shading and wind deflection resulted in cooler surfaces and moister soils than in the adjacent open desert in both the winter-spring growing season and in the summer dry season (Patten and Smith, 1980). These anticipated changes have led to speculation that shading of the surface would enhance the ability of an arid ecosystem to support growth of perennial grasses of commercial importance (Meinel and

Meinel, 1972). Such an environmental shift could have important ecological and economic implications.

A major reason for examining potential microclimatic alterations within large solar collector facilities is the 'fugitive dust' problem. Many proposed solar collector sites are in areas with high wind velocity during much of the year. Dust accumulation and sand blast will significantly lower the efficiency of mirrors (Berg, 1978) and flat-plate collectors (Garg, 1974), thus requiring the stabilization of ground surfaces in collector and mirror fields. There is a strong possibility that stabilization through artificial surfacing would be environmentally unacceptable for the large tracts of land anticipated to be utilized for solar collection. Asphalt paving would cause potential erosion problems on the periphery of the field, regional alterations in surface hydrology, and problems of land reclamation upon decommissioning of the facility. Gravel or similar surfacing is not expected to be effective in dust suppression over the long term. It appears that one of the most practical forms of surface stabilization may be revegetation.

Desert ecosystems with warm climates exhibit relatively low species diversity and community productivity, which often results in an ecosystem that is extremely fragile to any type of major disturbance. Although the southwestern United States and other arid regions have been dominated by deserts for centuries, many ecosystems have been modified within a short time period due to man's activities (Hastings and Turner, 1965). The Mojave Desert of southern California is a highly arid region in which the dry season coincides with the period of warmest temperatures. As a result, perennial plants of the region have adapted to the climate via either a xerophytic morphology combined with tolerance of extreme water stress, or with a drought-deciduous habit in which most of the transpiring surface is lost in the dry season (Bamberg et al., 1975). Ephemeral plant life is restricted almost entirely to the cool moist period due to their germination in response to predictable winter rains and to an acceleration of the life cycle to ensure completion of development in a very short period prior to the onset of hot, dry conditions (Beatley, 1967; 1974). It can be anticipated that desert plants of both perennial and ephemeral life form will respond to a less stressful surface microenvironment with greater productivity and by an extension of the period of active metabolism and growth into the dry season, presumably resulting in greater standing biomass and seed production. Indeed, these responses were observed in the native vegetation of the simulated site in Arizona (Patten and Smith, 1980). Moreover, a distinct invasion of both annual and perennial plants was observed onto the site such that the vegetation within the simulated heliostat field was both floristically and structurally dissimilar to that observed in the adjacent open desert.

A primary drawback of the simulation study conducted by Patten and Smith (1980) was that the site was too small to accurately simulate the surface microclimate anticipated to occur in a large operating facility. With the completion of Solar One, the opportunity to analyze vegetation responses in an operating facility became possible. However, prior to construction of Solar One, the entire site was cleared of existing vegetation, bladed, and compacted, so that the surface soil of Solar One may not be physically or chemically similar to an open undisturbed desert site. Ever so, a significant population of desert plants has annually germinated and become established within the heliostat field of Solar One, prompting us to initiate an investigation of the composition of the invading flora on the site, to determine its similarity to adjacent, relatively undisturbed desert vegetation, and to determine potential operational and safety impacts of the invading vegetation on normal plant operation.

2. Methods

Vegetation studies conducted at Solar One in 1983 and 1984 concentrated on three aspects of plant invasion of the site: (1) a floristic survey of the site; (2) quantification of density and aboveground biomass of the major species and vegetation as a whole; and (3) assessment of possible operational and safety hazards posed by the presence of vegetation on site. Two trips were made to the facility in each year, in late April and in early July in 1983, and in late March and early June in 1984. The timing of the two trips each year was in order to be on site at or near peak development of the spring and summer flora, respectively.

I initially intended to attempt a walk-through transect of the entire heliostat field to compile a complete floristic list of the invading flora on the site. However, consultations with site personnel and time constraints led to the conclusion that all activities would be restricted to the northeast (NE) quadrant of the field. The NE quadrant was chosen primarily because it is bordered by relatively undisturbed desert at its perimeter, whereas the other quadrants, particularly the two western quadrants, are bordered at their perimeters by disturbed habitats which support "old field" type vegetation or saltbush stands. Thus, seed rain into the NE quadrant may be more typical of what occurs in the open desert than in the other quadrants.

A complete walk-through of the NE quadrant was conducted during each site visit. Each new species encountered in the transect was collected and placed in a plant press to facilitate identification at a later date. An analogous transect was undertaken in the open desert adjacent to the NE quadrant. Plant identifications followed Munz (1968).

Quadrat-based quantitative analyses of the vegetation were conducted in the NE quadrant during each site visit to determine density and biomass of the vegetation as a whole and for each individual species. In April, 1983, four specific habitats were delimited for separate analyses: (1) the inner heliostat field, in the first six arcs near the tower; (2) the outer heliostat field, in the last six arcs at the periphery of the field; (3) an area within the fenced heliostat field area that had undergone similar surface preparation (i.e., had been bladed and compacted) but without overlying heliostats; and (4) an open desert control site outside the fence but adjacent to the NE quadrant of the field.

In each of the above four sites, ten 2.0 m radius circular quadrats were randomly placed in the site. In the field, heliostats were randomly chosen, and the central structural support of the heliostat served as the center of the circular quadrat; in the control and compacted control sites, quadrats were placed randomly along two line transects through the areas. Each circular quadrat was divided into four equal quarters by placing a 2 m line in each cardinal direction, and two 3.14 m^2 quadrats were examined at each location. Thus, 20 individual quadrats were analyzed in each site. All live, green plants within each quadrat were collected, identified to species, counted, and then separately bagged for later dry weight determinations, which were made after placing the plant materials at 60° C for 72 hours.

In early summer of 1983 it was decided by site personnel that the heliostat field needed to be cleared of all vegetation. As a result, I requested that certain groups of heliostats in the NE quadrant of the field be left uncleared in order to permit continuing observations of vegetation reestablishment in the heliostat field. Six such areas were selected , with each area containing a group of five heliostats. Two of the areas were in the inner part of the field and were contiguous, two were in mid-field, and two near the outer perimeter (Fig. 1). Thus, ten heliostats were chosen in each of three arcs of the field. The specific heliostats selected were:

Inner field: 0505, 0507, 0509, 0511, 0513, 0605, 0607,
0609, 0611, 0613, 0615

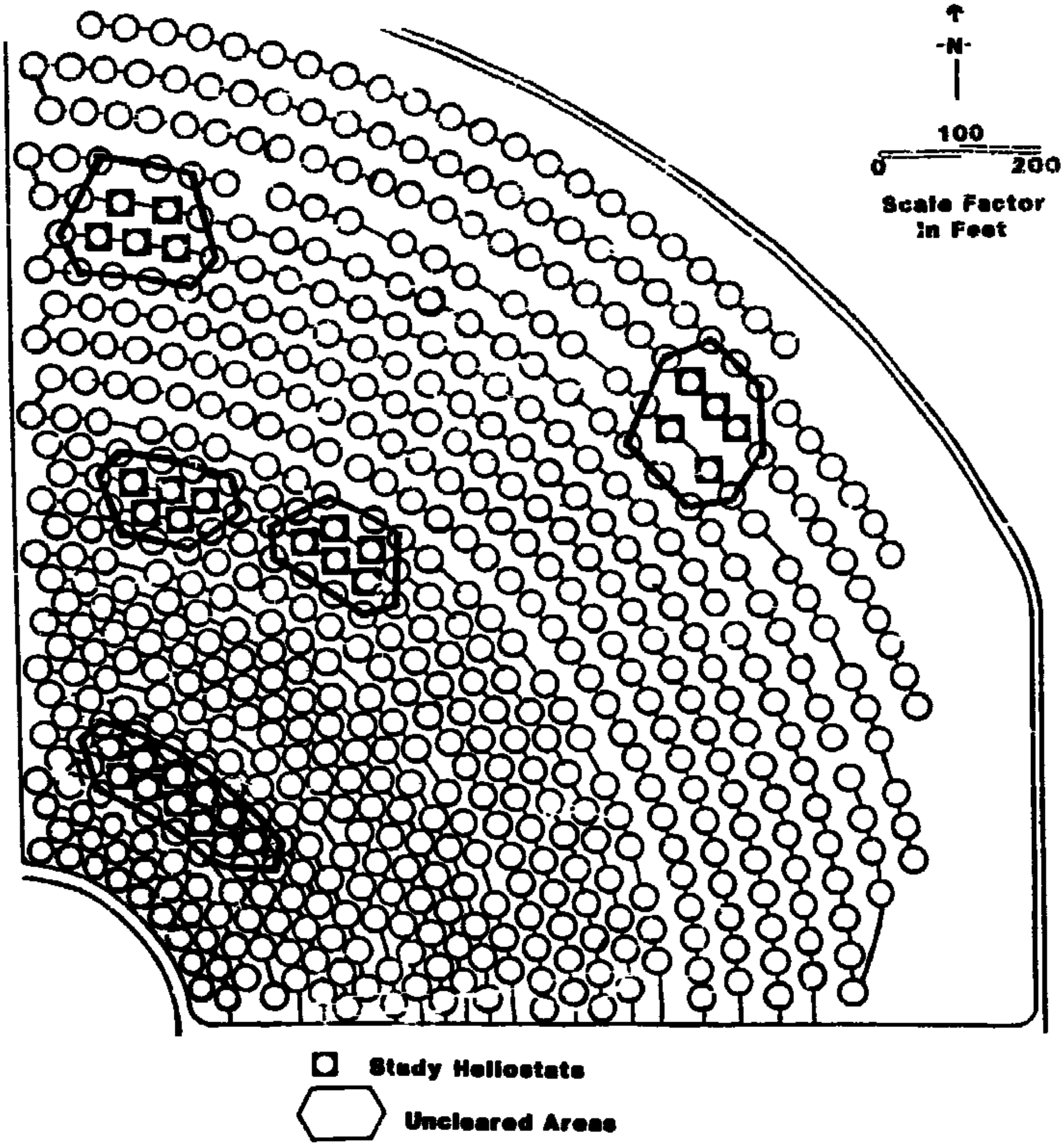

Fig. 1. Study Heliostat Locations in NE Quadrant

Mid-field: 1607, 1609, 1705, 1707, 1709, 1715, 1717, 1719, 1817, 1819

Outer-field: 2403, 2405, 2407, 2505, 2507, 2635, 2637, 2743, 2745, 2747

The uncleared area around each group of heliostats extended to the next adjacent heliostat around the perimeter of each heliostat group. The selected heliostats were identified in the field by wrapping their pedestals with orange surveyor's tape.

In early July, 1983 and in 1984, quadrat-based analyses were conducted at the designated heliostats in the NE quadrant and in the adjacent open desert control site. A 2.0 m radius circular plot was placed beneath each heliostat, and the circular plot was further subdivided into four subplots based on the cardinal directions. At each heliostat two subplots were analyzed, so that a survey of twenty 3.14 m^2 area quadrats was conducted. In each plot, all plant material was collected, separated by species, counted, and bagged. In the summer samples two species, Salsola iberica (Russian thistle) and Atriplex polycarpa (saltbush), were too large and/or numerous to make whole plant collections feasible. As a result, a dimension analysis procedure was undertaken in which linear measurements of height and width of each plant canopy taken in the field could be converted to canopy volume and then to canopy biomass (e.g., Ludwig et al., 1975). The canopy of each species was approximated by an upper half spheroid, so that canopy volume could be derived from height and width measurements by the equation:

$$V = 1.33 \pi r^2 h$$

where V is canopy volume (in dm^3) and h and r the height and radius of the canopy (in dm). An off-site harvest of a size series of each species then allowed regression equations to be formulated which related canopy volume (in dm^3) to aboveground dry weight biomass (in g), specifically:

Salsola iberica, $B = 1.03V + 1.5$ ($r^2 = .977$, $n = 20$), and

Atriplex polycarpa, $B = 1.65V + 6.0$ ($r^2 = .988$, $n = 15$).

As a result of this analysis, it was possible to quantify the average height and overall cover of *Salsola*, the dominant invading plant on the site in the summer. Individuals of *Salsola* and *Atriplex* which were smaller than 1 x 1 dm were harvested, counted, and bagged for later biomass determination.

Only a limited amount of data was collected in the two site visits in 1984. In the March visit, high winds allowed only several hours of access into the heliostat field each morning. Between the March and June visits herbicide was apparently applied throughout the heliostat field, resulting in almost complete mortality of all dicotyledonous plant species. Only the grasses were unaffected by the treatment. Thus, comparable analyses as conducted in 1983 were not possible in 1984, resulting in a very limited data set in 1984.

3. Results and Discussion

3.1 Winter-spring vegetation

A floristic list of the plant species occurring in the NE quadrant of the heliostat field and in the adjacent control site is given in Table 1 for the spring site visits. Twenty-seven (27) vascular plant species were collected, representing 21 genera and 10 families of flowering plants. No cryptogams (i.e. ferns, mosses, fungi or lichens) were observed on the site, although some dead fungi were observed; P. Flanagan (personal communication) reported that abundant numbers of fungi occurred on site in 1982, particularly in the NW quadrant of the heliostat field. Four species of grasses occurred in the heliostat field, but all were annual introduced species. Of the dicots found in the heliostat field, most appeared to be weedy introduced species, particularly composites such as Sonchus spp. and crucifers such as Brassica, Descurainea and Sisymbrium. Most of these weedy introduced dicots were not found in the open desert. Two species found in the open desert, Hesperocallis undulata and Tiquilia plicata, are native perennial forbs, and are considered non-weedy. Thus, the spring vegetation of the heliostat field could be considered weedy in nature based on its floristic composition.

Quantitative estimates of plant density and biomass in April, 1983 are given in Table 2. Due to a dense germination of Schismus arabicus, the open desert control site had the greatest overall plant density and biomass of the four sites. Limited data from March, 1984 shows similar results, with total plant density being 1,65[illegible] per m^2 in the control vs only 81 per m^2 in the heliostat field, while aboveground biomass was 33.6 and 4.5 g m^{-2} in the control and heliostat field, respectively.

Table 1. Planta taxa collected at Solar One during the spring of 1983 and 1984. Site designations refer to taxa which were restricted to the control site (C) or to the heliostat field (H); other taxa occurred in both locations.

Family	Species (common name)	Site
MONOCOTYLEDONAE		
Gramineae	*Bromus rubens** (Foxtail Chess)	
	*Hordeum leporinum**	
	*Schismus arabicus**	
	*S. barbatus**	
Liliaceae	*Allium* sp. (Onion)	
	Hesperocallis undulata (Desert Lily)	
DICOTYLEDONAE		
Boraginaceae	*Amsinckia tesselata* (Fiddleneck)	
	Cryptantha angustifolia	
	C. sp.	
	Pectocarya recurvata	
	Tiquilia plicata	C
Chenopodiaceae	*Atriplex hymenelytra* (Desert Holly)	H
	A. polycarpa (Saltbush)	
	Chenopodium atrovirens	H
	C. sp.	
	*Salsola iberica** (Russian Thistle)	
Compositae	*Malacothrix glabrata* (Desert Dandelion)	
	*Sonchus asper** (Sow Thistle)	H
	*S. oleraceus** (Spiny Sow Thistle)	H
Cruciferae	*Brassica nigra** (Black Mustard)	H
	*Descurainea sophia** (Tansy Mustard)	H
	*Sisymbrium irio** (London Rocket)	
Geraniaceae	*Erodium cicutarium** (Filaree)	
Malvaceae	*Malvastrum exile*	H
Hydrophyllaceae	*Phacelia crenulata*	H
Polygonaceae	*Eriogonum trichopes*	
	E. sp.	

*Indicates introduced taxa.

Table 2. April 1983 vegetation analysis of four sites at Solar One. Site abbreviations are: CC = compacted control; OH = outside heliostat field; and IH = interior heliostat field. Data are expressed as mean ± 1 S.E.

Parameter	Control	CC	OH	IH
Total plant density (no. m^{-2})	2,116 ± 185	35.7 ± 5.0*	25.8 ± 3.7*	20.0 ± 2. 2*
Grass density (no. m^{-2})	2,109 ± 184	35.6 ± 5.0*	24.9 ± 3.6*	17.6 ± 2.3*
Herbaceous dicot density (no.m^{-2})	6.9 ± 1.6	0.09 ± 0.04*	0.87 ± 0.16*	2.35 ± 0.31*
Total plant biomass (g m^{-2})	46.2 ± 6.7	1.98 ± 0.4*	2.61 ± 0.47*	3.38 ± 0.25*
Grass biomass (g m^{-2})	45.1 ± 6.2	1.97 ± 0.4*	2.44 ± 0.44*	2.63 ± 0.36*
Herbaceous dicot biomass (g m^{-2})	1.10 ± 0.47	0.004 ± 0.001*	0.17 ± 0.05*	0.75 ± 0.29
Species Richness (no. $quadrat^{-1}$)	3.6 ± 0.35	1.9 ± 0.14*	4.9 ± 0.32*	8.8 ± 0.36*
Species Diversity (H')	.009 ± .034	.010 ± .002	.438 ± .063*	.760 ± .110*

*Significantly different from control site at P = .05.

The similarity of plant density and biomass in the three sites within the heliostat field area (Table 2) suggests that previous surface clearing and disturbances was the primary factor limiting vegetation presence in the heliostat field. However, the greater numbers and biomass of herbaceous dicots, plus a greater species richness and diversity in the two heliostat field sites relative to the compacted control site suggests that the vegetation was already showing a response to the less stressful microclimate beneath the heliostats. This is further illustrated by a comparison of the inner and outer heliostat fields, where more herbaceous dicots and a higher species diversity occurred in the inner field, where heliostat packing is higher and thus shading is more complete. Even though species richness in the control site was comparable to that in the outer heliostat field, control species diversity was very low due to the high relative density of Schismus arabicus in the control site. The relatively depauperate vegetation cover and low diversity of plants in the compacted control site illustrates that surface disturbance alone was a negative factor in vegetation development on the heliostat site.

A breakdown of the overall vegetation into representative species further illustrates the differences between the vegetation structure and composition of the heliostat field and control sites (Table 3). Schicmus arabicus dominated the two control sites to a much greater extent than in the heliostat field proper. The similarity of species composition and relative importance between the control and compacted control sites

Table 3. Importance values of major species form the April 1983 vegetation survey of Solar One. Site abbreviations are as in Table 2. Importance value (IV) is the sum of relative frequency (RF), relatively density (RD), and relative biomass (RB), and thus totals 300 for each site.

Species	Control				CC				OH				IH			
	RF	RD	RB	IV	RF	RD	RB	IV	RF	RD	RB	IV	RF	RD	RB	IV
Grasses:																
<u>Schismus arabicus</u>*	44	99.6	98	242	69	99.8	99.8	269	34	95	89	218	218	86	64	168
<u>Bromus rubens</u>	4.4	0.1	<0.1	4.5	3.4	<0.1	<0.1	3	4.3	1.0	2.1	7	7	0	0	0
<u>Hordeum leporinum</u>	4.4	<0.1	<0.1	4.5	0	0	0	0	8.5	0.5	1.8	11	7.1	3.1	6.8	17
Forbs:																
<u>Brassica nigra</u>	0	0	0	0	0	0	0	0	5.6	0.2	1.2	7	1.8	0.1	11	13
<u>Cryptantha angustifolia</u>	16	0.3	0.3	16	14	0.1	0.1	14	16	1.6	1.5	19	13	3.2	1.7	18
<u>Descurainea sophia</u>	0	0	0	0	3.4	<0.1	<0.1	3	8.5	0.5	2.0	11	11	0.7	0.8	13
<u>Eriogonum trichopes</u>	2.2	<0.1	<0.1	2	0	0	0	0	1.4	<0.1	<0.1	1	1.8	0.2	0.1	2
<u>Erodium cicutarium</u>	11	<0.1	<0.1	13	0	0	0	0	4.3	0.1	0.6	5	11	0.7	1.3	13
<u>Phacelia crenulata</u>	0	0	0	0	0	0	0	0	2.7	0.1	<0.1	3	13	0.7	0.1	14
<u>Salsola iberica</u>	6.7	<0.1	<0.1	7	6.9	0.1	<0.1	7	13	0.8	1.1	15	13	4.7	2.1	20
Others	11	0.1	0.3	11	3.5	<0.1	<0.1	4	4.7	0.1	0.3	5	10	0.8	12	23

*Includes some individuals of <u>Schismus barbatus</u>

suggests that the occurrence of weedy sub-dominants in the heliostat field, such as Brassica, Descurainea and Phacelia, was due to shading and presumed higher soil moisture content beneath heliostats, rather than due to disturbance per se. Otherwise, we should expect these species to occur in the compacted control site in equal quantities if the heliostat field vegetation is interpreted as strictly a disturbance related flora. Although Schismus was still the dominant species in the heliostat field, the greater presence of other species and the relative evenness of the importance of each of these species, particularly in the inner heliostat field, helps explain the higher species diversity noted previously in the heliostat field, especially the inner field. It is an accepted premise in plant ecology that species diversity is negatively correlated to the stressfulness of the environment, i.e., more stressful habitats tend to have lower species diversity (e.g., Danin, 1976; 1978). Assuming a less stressful surface micro-environment occurs beneath each heliostat due to shading and possibly added moisture, and that the inner field would be anticipated to be less stressful than the outer field due to greater heliostat density, then the vegetation data obtained here supports that general premise.

Average individual biomass and the percentage of aboveground biomass allocated to reproductive structures (i.e., flowers and seeds) in the major species in April, 1983 is given in Table 4. A clear trend exists in which most of the species were much larger on an individual basis in the heliostat field relative to the control site. In March, 1984 the average individual aboveground biomass of Schismus arabicus was 61 mg in the

Table 4. April 1983 vegetation analysis at Solar One. Site abbreviations are: OH = outside heliostat field; IH = interior heliostat field.

Parameter	Control	OH	IH
Total plant density ($n \cdot m^{-2}$)	2,116	26*	20*
Grass density ($n \cdot m^{-2}$)	2,109	25*	18*
Forb density ($n \cdot m^{-2}$)	6.9	0.9*	2.4*
Total plant biomass ($g \cdot m^{-2}$)	46	2.6*	3.4*
Grass biomass ($g \cdot m^{-2}$)	45	2.4*	2.6*
Forb biomass ($g \cdot m^{-2}$)	1.1	0.2*	0.75
Species diversity (H^1)	0.01	0.44*	0.76*

*Significantly different from control site ($\underline{p}=0.05$)

heliostat field and 9 mg in the control site. The lower average plant biomass in the control site may be attributed to intense competition due to the very high plant density on the site, although rapid development of the control plants in order to ensure seed set prior to the onset of water stress induced senescence (e.g., Beatley, 1967) may also have been a causal factor in their small size.

Differences in plant size were generally not significantly different between the two heliostat field sites in 1983, except for the ubiquitous Schismus which attained maximum size in the inner field. In several of the sub-dominant species in which the largest plants were observed in the outer field, the plants of that species appeared to be developmentally delayed in the inner field, possibly due to germination occurring later in the year in the inner field. This interpretation is partially borne out by noting that reproductive allocation was usually lowest in the inner field. Thus, those plants in the inner field are interpreted as having been at an earlier stage in their respective developmental cycles, rather than having an intrinsically lower seed production due to the shaded conditions beneath the heliostats. Smith (1981) similarly found the life cycles of plants beneath simulated collectors to be delayed in time, but to still have an equal reproductive allocation at peak development as did plants in the open. Thus, this data, although certainly not conclusive, indicates that plants in shaded microsites beneath heliostats at Solar One may have an extended growth period into the dry late spring due to a reduction in the onset of water stress and high surface temperature.

3.2 Summer vegetation

The plant taxa collected at Solar One in July, 1983 are given in Table 5. Due to herbicide treatment of the field and resulting absence of dicotyledonous plants, a comparable analysis could not be conducted in 1984. A distinct summer flora was observed in 1983 although certain winter-spring species such as *Schismus* spp. and *Cryptantha* sp. were observed in the heliostat field. The dominant plant in 1983 was the ubiquitous summer weed *Salsola iberica* (Russian thistle), which occurred abundantly in the heliostat field and in disturbed habitats in the open desert. Other introduced weedy species such as *Conyza canadensis*, *Lactuca serriola*, and *Sonchus oleraceus*, were found to occur only within the heliostat field. *Ambrosia acanthicarpa*, which is a common weed native to the area, occurred abundantly in the heliostat field only.

Several shrubby perennials had also become established in the heliostat field by July, 1983, the most common being *Atriplex polycarpa* (saltbush), which was the dominant shrub on the heliostat field site prior to it being cleared (Turner, 1982). Other shrubby plants observed in the heliostat field but not in the open desert were *Atriplex hymenelytra*, *Bebbia juncea*, and *Eriogonum plumatella*. None of these three species were observed in the vicinity of Solar One; all are characteristically found in habitats such as washes or stony slopes of the Mojave Desert

Table 5. Plant taxa collected at Solar One in July 1983. Site designations are as in Table 1.

Family	Species (common name)	Site
MONOCOTYLEDONAE		
Gramineae	_Oryzopsis hymenoides_ (Indian Ricegrass)	H
	_Schismus arabicus_*	
	_S. barbatus_*	
DICOTYLEDONAE		
Boraginaceae	_Cryptantha_ sp.	H
	Tiquilia plicata	C
Chenopodiaceae	_Atriplex hymenelytra_ (Desert Holly)	H
	A. polycarpa (Saltbush)	
	Chenopodium incanum	H
	_C._ sp.	H
	_Salsola iberica_* (Russian Thistle)	
Compositae	_Ambrosia acanthicarpa_ (Sand-bur)	H
	Bebbia juncea (Sweet Bush)	H
	_Conyza canadensis_* (Horseweed)	H
	_Lactuca serriola_* (Prickly Lettuce)	H
	_Palafoxia linearis_*	H
	Psathyrotes ramosissima	C
	_Sonchus oleraceus_* (Sow Thistle)	H
Malvaceae	_Malvastrum exile_	H
Plantaginaceae	_Plantago lanceolata_* (Ribgrass)	H
Polygonaceae	_Eriogonum inflatum_ (Desert Trumpet)	C
	E. plumatella	H
	E. trichopes	
Solanaceae	_Datura meteloides_ (Jimsonweed)	C

*Indicates introduced taxa.

(Munz, 1968), and are thus not typically found in the creosotebush flats which characterize the vicinity of Solar One. Again, the presence of these species is an indication of the more moderate environment occurring within the heliostat field. A tufted perennial grass, Oryzopsis hymenoides, was also observed in the heliostat field in 1983, whereas only introduced annual grasses had been observed earlier in the spring. Several native perennial forbs, such as Eriogonum inflatum, and Datura meteloides, were found in the control site but not in the heliostat field. Thus, as in the spring, the flora which was restricted to one site or the other could be generally characterized as introduced, weedy annuals in the heliostat field and native perennial forbs in the open desert. It should also be noted that the dominant perennial plant of the control site, Larrea tridentata (creosotebush), had not yet exhibited any germination or establishment in the heliostat field, although Atriplex polycarpa was becoming well established.

Quantitative estimates of overall density and aboveground biomass of the vegetation on each site in 1983 is given in Table 6. By July, essentially all ephemeral vegetation had died in the open desert. In contrast, green vegetation was quite abundant in the heliostat field and steadily increased in both numbers and total biomass into the inner field. Although the vegetation in all sites was composed of predominantly annual plants (e.g., Salsola), perennial plants were also represented. Of interest is that the shrubby perennials in the heliostat field were most abundant in the mid-field location. It is possible that seed material of these shrubby plants was more abundant

Table 6. July 1983 vegetation analysis of four sites at Solar One. Site abbreviations designate the outside (OH), middle (MH) and interior (IH) heliostat fields. Data are expressed as mean ±1 S.E. All parameters in each heliostat site were significantly different from the control site at the 1% level.

Parameter	Control	OH	MH	IH
Total plant density (no. m^{-2})	0.05 ± 0.02	2.18 ± 0.26	5.35 ± 0.3	10.9 ± 0.9
Annual plant density (no. m^{-2})	0.05 ± 0.02	2.01 ± 0.23	4.75 ± 0.3	10.7 ± 0.9
Perennial plant density* (no. m^{-2})	0.004 ± 0.01	0.17 ± 0.04	0.58 ± 0.24	0.18 ± 0.03
Total plant biomass (g m^{-2})	0.36 ± 0.17	27.25 ± 10.6	39.0 ± 11.0	82.9 ± 10.9
Annual plant biomass (g m^{-2})	0.34 ± 0.16	26.6 ± 10.3	36.4 ± 10.6	82.0 ± 11.0
Perenial plant biomass (g m^{-2})	0.02 ± 0.01	0.62 ± 0.41	2.64 ± 1.35	0.92 ± 0.24
Species Richness (no. $quadrat^{-1}$)	0.30 ± 0.12	1.93 ± 0.24	3.44 ± 0.23	4.25 ± 0.35
Species Diversity (H')	0.039 ± 0.02	0.381 ± 0.086	0.549 ± 0.080	0.303 ± 0.22
Salsola iberica Cover (% surface cover)	0.09 ± 0.04	4.55 ± 1.54	6.6 ± 1.6	14.4 ± 1.55

*Does not include mature shrubs in control site, only young plants.

toward the exterior of the field (i.e., nearer the seed source) and that invasion by perennials may be delayed into the inner field. In contrast, the inner field supported the highest density and biomass of weedy annuals, and the highest cover of Salsola. Thus, it would appear that weedy plants readily penetrated into the inner field. Both species richness and diversity were highest in the heliostat field, with richness being highest in the inner field and diversity highest at the mid-field location. This discrepancy was due to the overall dominance of Salsola in the inner field vegetation, resulting in a reduced evenness of the vegetation structure.

A comparison of the summer (Table 6) and spring (Table 2) vegetation analyses in 1983 shows that plant standing biomass in the heliostat field was roughly 20-fold greater in the summer than in the spring, and was roughly equivalent in the summer to the Schismus dominated vegetation in the control site in the spring. This difference in biomass between spring and summer vegetations is illustrated in Fig. 2. However, species diversity and richness were highest in the heliostat field in the spring, again due to the dominance of the vegetation by Salsola in the summer. The shift in green vegetation biomass from being greatest in the open desert in the spring to being greatest in the inner heliostat field in the summer clearly shows the importance of microclimate in determining potential biomass production in the different sites. In the cool winter-spring period, germination and eventual standing biomass was highest in the open desert, where full sunlight would not be detrimental due

Fig. 2. Plant invasion into Solar One heliostat field in April (above) and July (below) 1983. Vegetation in April is predominantly Schismus arabicus, while the large plants in July are Salsola iberica.

to moist soils and cool surface temperatures. In the dry, hot summer, maximum plant development occurred in the most heavily shaded sites, and progressively decreased as the amount of sunlit area increased. Observations in the field in July, 1983 showing most of the green, viable plants to be clustered around heliostat bases further illustrates this (Fig. 3).

An analysis of species importance in each site shows Salsola iberica had the highest IV in each site, although Schismus spp. were still abundant in the heliostat field (Table 7). The greater importance of Salsola in the inner field was attributed not only to a greater overall standing biomass, but also to a higher density of plants than in the other sites. This higher density was mainly due to a large number of Salsola seedlings which occurred in the inner field, indicating that the total biomass and cover of Salsola in the inner field would be expected to be considerably higher at peak development later in the summer. Several species, such as Ambrosia acanthicarpa, Bebbia juncea and Lactuca serriola, showed highest IV in the inner heliostat field, whereas others, such as Atriplex polycarpa and Cryptantha sp., showed maximum development in the outer and mid-field locations.

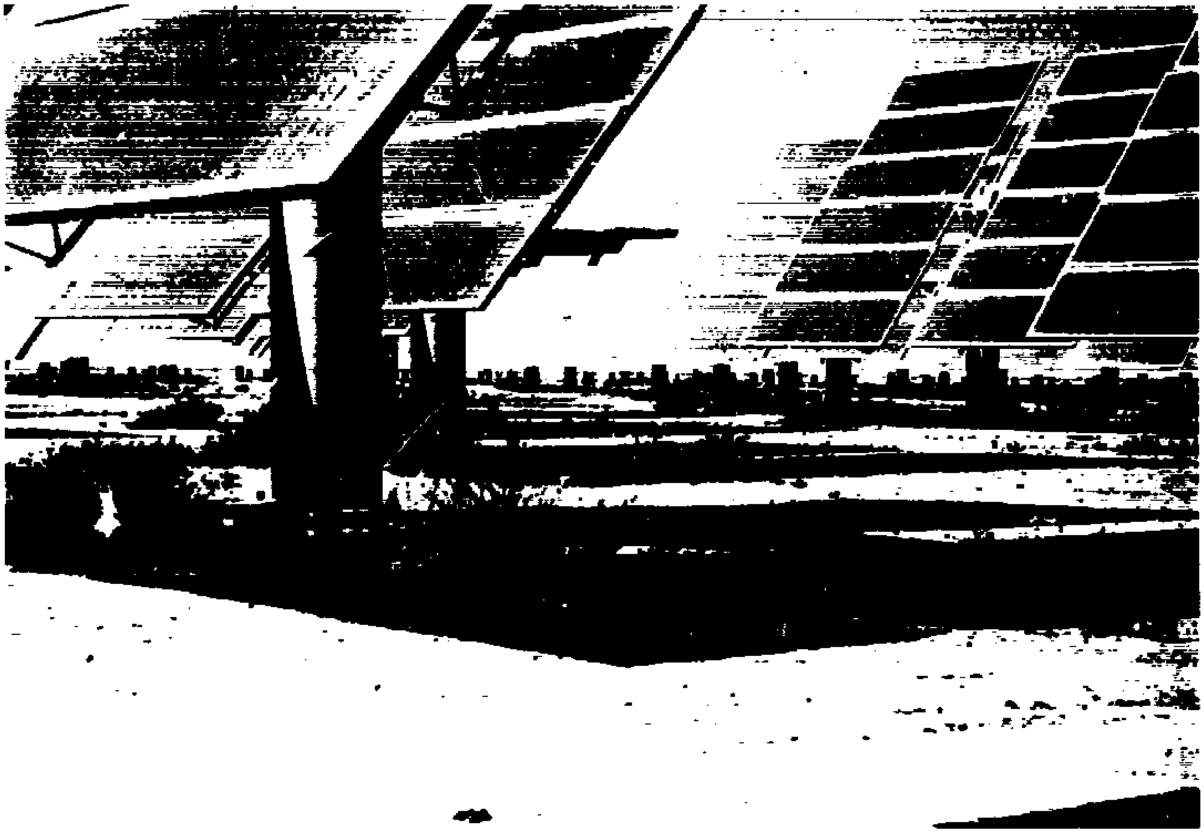

Fig. 3. July 1983 vegetation in the Solar One heliostat field showing clustering of plants around heliostat pedestals.

Table 7. Importance values of major species from the July 1983 vegetation survey of Solar One. Site abbreviations are as in Table 6. Parameter abbreviations are as in Table 3.

Species	Control				OH				MH				IH			
	RF	RD	RB	IV	RF	RD	RB	IV	RF	RD	RB	IV	RF	RD	RB	IV
Salsola iberica	40	62	83	185	32	9	92	133	29	25	86	140	24	45	94	163
Schismus spp.	0	0	0	0	36	86	5	127	26	61	3	90	19	46	3	58
Ambrosia acanthicarpa	0	0	0	0	4	0.2	0.1	4	16	2.6	1.7	20	18	4.4	1.3	24
Atriplex polycarpa	0	0	0	0	8	1	3	12	7	0.9	6	14	3.5	0.3	0.3	4
Cryptantha sp.	0	0	0	0	14	3	0.4	17	14	10	3	27	3.5	0.3	0.1	4
Lactuca serriola	0	0	0	0	0	0	0	0	1.7	0.2	<0.1	2	9	1.5	0.8	11
Bebbia juncea	0	0	0	0	0	0	0	0	0	0	0	0	9	0.9	0.8	11
Eriogonum trichopes	60	38	17	115	0	0	0	0	0	0	0	0	0	0	0	0
Other Species	0	0	0	0	6	1	0.1	7	6	0.7	0.1	7	15	1.5	0.1	17

Average aboveground biomass of the major species in each site in 1983 is given in Table 8. Individuals of Salsola iberica developed into much larger plants in the heliostat field than in the open desert, and reached maximum mean size in the outer field. However, the reduced mean size in the inner and mid-field sites was due to the larger number of small seedlings which occurred there, as the largest individual Salsola plants were actually encountered in the inner field. This apparent later germination in the inner field is well illustrated by the average biomass data for Atriplex polycarpa, as mature individuals of this species were only observed in the outer and mid-field locations. In contrast, Schismus spp., which germinated in late winter, displayed maximum individual biomass in the inner field. Furthermore, an analysis was made in July of the percentage of Schismus plants which still displayed a green, live canopy. There was less than 1% live Schismus in the control site, 28% live in the outer field, and fully 59% live in the inner field. Limited observations in 1984 similarly showed enhanced Schismus survival in the heliostat field vs no green plants in the control site in late May. The combination of larger plant size and increased survival into the dry season again points to the potential of desert species to exploit more favorable habitats in the shade of heliostats than what is generally available to them in the open desert.

Table 8. Average plant biomass (as dry weight per individual) of the major species at Solar One, July 1983. Site abbreviations are as in Table 6.

Species	Control	OH	MH	IH
Salsola iberica	6.5	65.0*	31.1*	27.3*
Schismus spp.	---	0.37	0.38	0.57
Ambrosia acanthicarpa	---	2.1	3.9	2.7
Atriplex polycarpa	---	14.1	22.4	2.2
Crytantha sp.	---	1.0	1.7	3.9
Lactuca serriola	---	---	0.7	6.9
Bebbia juncea	---	---	---	8.5

*Significantly different from control at P = .05.

4. Effects of Stow Position and Heliostat Washing.

Heliostat stow position (i.e., at night and during inclement weather) and washing schedules may have important influences on vegetation spatial structure beneath each heliostat. Heliostats are generally stowed horizontally with the mirror surfaces facing downward, or vertically with the mirror surface facing away from the prevailing wind source (Fig. 4). When vertically stowed, a "drip line" can be created along the length of the heliostat during rainfall events. This is also the stow position used to wash each mirror, as a large truck with a water hose is utilized to wash each row of heliostats on a semi-regular basis, as shown in Figure 5. This potentially concentrated source of water, particularly from heliostat washing, can result in a dense band of vegetation along the "drip line" (Fig. 6). Although not specifically observed in the summer of 1983, this zone could become dominated by large <u>Salsola</u> and possibly <u>Atriplex</u> bushes during the summer months. Since the apparently preferred stow position is vertical, this band of vegetation would be expected to be a distinct feature of the heliostat field vegetation if left unaltered.

5. Conclusions of Vegetation Inventories

A number of conclusions can be drawn from the floristic inventories and quantitative vegetation analyses conducted in 1983-84 at Solar One. Quantitatively, it was found that plant population density and standing biomass were higher in the open desert than in the heliostat field in the spring, but were substantially higher in the heliostat field in the summer. An

Fig. 4. Horizontal (above) and vertical (below) heliostat stow positions utilized at Solar One.

Fig. 5. Heliostat washing technique at Solar One. Heliostats are stowed vertically by row and each mirror is washed using a high pressure hose. Wash water is not collected and so it enters the soil at the base of each heliostat.

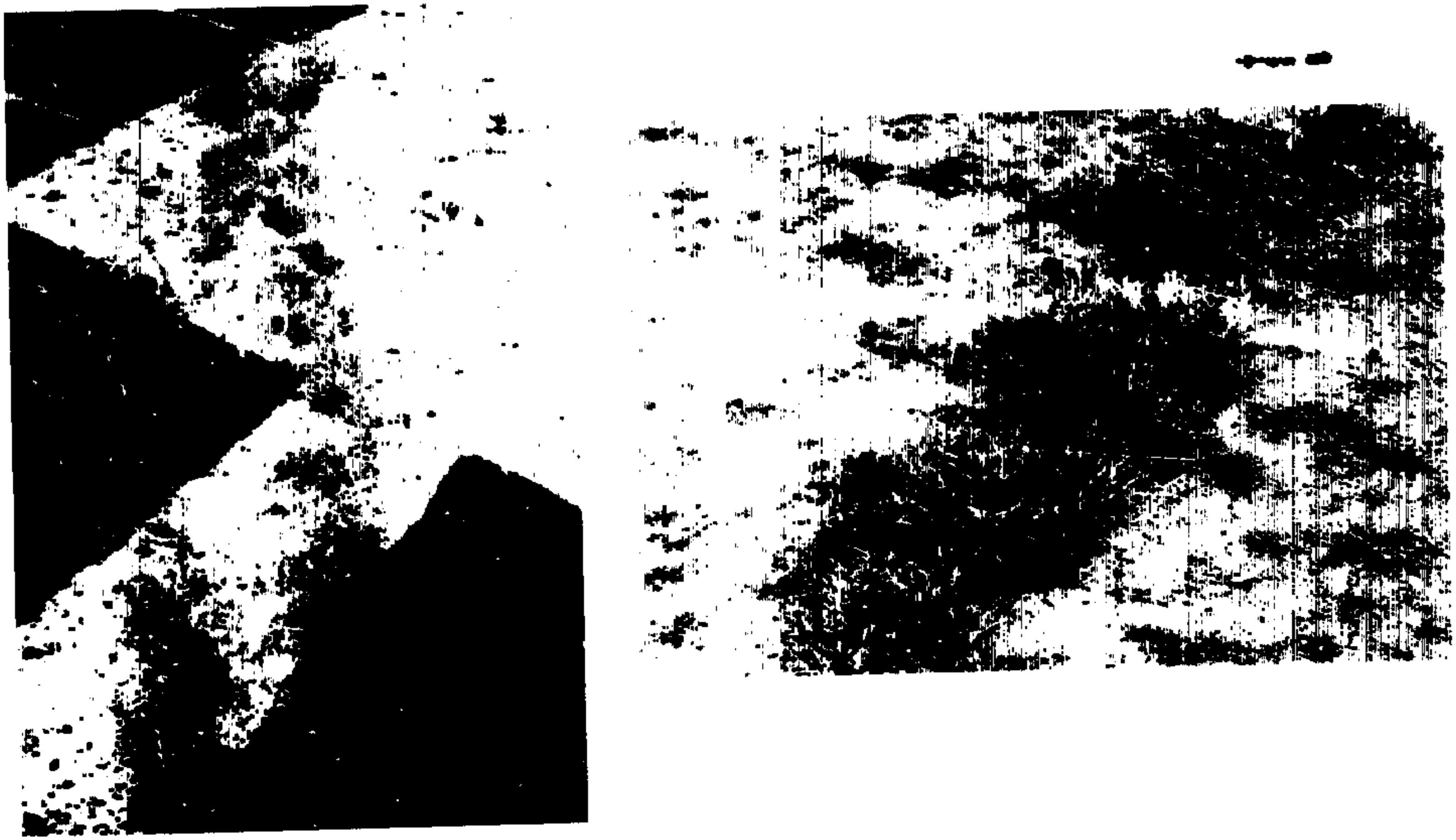

Fig. 6. Band of vegetation formed along the "drip line" at the base of each heliostat, taken in March, 1984 (left). This "drip line" apparently runs along the base of each heliostat when it is stowed vertically. At right is a close-up of the vegetation band, which is mainly composed of Schismus arabicus.

annual introduced species dominated the vegetation of both sites at both times of the year; the grass [illegible]mus *arabicus* in the spring and the weed *Salsola iberic* [illegible] the summer. Species richness and diversity were higher in the heliostat field in both spring and summer, and average plant size of individual species also tended to be greater in the heliostat field in both seasons. If the dominant species is excluded from consideration in both the spring and summer floristic inventories, a distinct dissimilarity of the flora and vegetation structure can be seen between the two sites. In general, the heliostat field was characterized by introduced weedy annuals which did not tend to occur in the open desert, whereas several species of native perennial forbs characterized the open desert flora, but were conspicuously absent from the heliostat field.

Analysis of an area within the fenced heliostat field which had been cleared and compacted but did not have overlying heliostats in April, 1983 pointed to two conclusions regarding plant invasion into the site: (1) the low plant density and diversity there indicated that disturbance alone did not enhance vegetation establishment into the heliostat field, and (2) the lack of similarity in floristic composition between the compacted control area and actual heliostat field sites indicate that the higher diversity and common occurrence of weedy annuals, many of them quite mesophytic in character, must be attributed to the presence of the overlying structures in the heliostat field and possibly to added moisture from heliostat washing. Although quantitative data was not collected from the compacted control area in the summer, observations indicate it was again quite similar in species composition to the open desert rather than to the heliostat field.

The results obtained in this study clearly show that plant development has been greatly enhanced in the more moderate microenvironments created beneath heliostats. Three factors may account for this: (1) reduced surface temperature extremes due to shading by the heliostats, (2) reduced evaporation from the soil surface, and thus conservation of soil moisture, due to shading and wind deflection by the heliostats, and (3) addition of water to the soil beneath each heliostat due to regular washing of the mirror surfaces. Because it was not possible to obtain accompanying microclimatic and soil moisture data in this study, the relative importance of these factors cannot be addressed. Probably all three play a role in stimulating larger plant size, greater overall plant cover and enhanced survival into the dry season, as was observed beneath heliostats in this study.

6. Operational and Safety Aspects

Because the heliostat field at Solar One has been cleared of vegetation in each year of operation, it can be concluded that the presence of vegetation on site has posed operational/safety problems or hazards. Our observations in 1983 and discussions with site personnel indicate that dense vegetation cover can affect normal heliostat operation and may by potentially deleterious to worker safety. It appears that the main problem has been the presence of dense stands of tumbleweed, Salsola iberica, which occur in the summer months. Tumbleweeds tend to occur most abundantly on the side of the heliostat pedestal which faces the central receiver and toward which the mirror surface usually faces (Fig. 7). This is also the side of the pedestal where the control box is located, so that access to the control box for routine maintenance or testing activities may be difficult beneath many heliostats.

There is, as yet, no indication of plants actually interfering directly with the normal tracking motion of the heliostats, although spring and summer weeds were observed to have grown to a meter in height. The largest plants in the spring flora were Brassica nigra and Sonchus oleraceus, neither of which occurred in dense enough stands to pose a major problem. In July, however, there were several large tumbleweeds (each larger than 50 x 50 cm) beneath almost every heliostat in the inner field of the NE quadrant, and tumbleweeds were even more numerous in the western quadrants of the heliostat field, where Salsola cover appeared to be almost complete beneath some heliostats. A particular problem with tumbleweeds is their tendency to become uprooted after dying and then blow across the site during windstorms, often becoming accumulated in large "drifts" against fences or other structures. Prevention of this dispersal of dead tumbleweeds was a possible reason for clearing them from the site while they were sill young and green.

Worker safety aspects of vegetation presence is primarily related to the potential shelter that vegetation may provide for animal life, particularly poisonous snakes. There have been

Fig. 7. Cluster of tumbleweeds (Salsola iberica) around heliostat pedestal base in July, 1983 at Solar One. Note top of control box, which is almost completely overgrown by the tumbleweeds.

numerous sightings of rattlesnakes in the Solar One heliostat field, even in the central area near the tower. As of yet, no personnel have been bitten, but the presence of dense vegetation in the vicinity of heliostats and particularly accumulated around control boxes increases the risk. Most snake sightings have been at night or near sunrise and sunset, so the snakes undoubtedly have access to shelter in the form of burrows or protected areas independent of vegetation cover. Thus, removal of vegetation alone will not solve the potential problem. Nevertheless, it will reduce the cover available for snakes and possibly for the rodents they feed on, and thus possibly reduce their presence in the heliostat field.

With the exception of the dense tumbleweed population, vegetation does not appear to have yet posed serious problems in the normal operation of Solar One. Vegetation cover may be somewhat beneficial in that once it attains a reasonable amount of cover, it will act as a surface stabilizer and thus reduce the generation of fugitive dust within the field. Indications are that dust accumulation and possibly sand blast reduces the efficiency of the heliostats at Solar One, so that a reduction of fugitive dust formation within the field may be beneficial. However, the degree to which plant cover may potentially reduce fugitive dust formation in the field is not known, and the stabilization of surfaces with plant species other than tumbleweed would clearly be the desired result. Thus, a recommendation would be to clear only problem plant species such as tumbleweed and some of the large weedy spring annuals such as black mustard and sow thistle, and to leave in place obvious perennial (i.e. woody) plant species such as saltbush and other desert shrubs. Since all clearing efforts have been by mechanical means, this strategy would not be difficult to implement.

7. Future Research Opportunities

The composition and successional trends of the vegetation at Solar One may have important applications toward how surfaces are dealt with in future, larger facilities. Our preliminary discussions with SCE personnel indicate that an attempt may be made at Solar 100 to preserve islands of natural vegetation, clearing only corridors for vehicular traffic and heliostat placement. If this turns out to be feasible and is undertaken, then the results of this preliminary revegetation study may have particular relevance. Ideally, it is hoped that at some future date it will be possible to do a thorough analysis of plant responses in the heliostat field of an operating solar facility, possibly using pre-selected transplant material. In such a study, a complete accompanying microclimatic analysis of the various microenvironments within the field would allow definitive conclusions to be made relating plant responses to habitat modifications imposed by the shading and wind deflection of the heliostat array. The long term goal of such a project would be to select plant species which would readily exploit the heliostat field microclimate, provide adequate soil stabilization throughout the year, and not contribute operational and/or safety problems to the facility in question.

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Effects of artificially imposed shade on a Sonoran Desert ecosystem: microclimate and vegetation

S. D. Smith*†, D. T. Patten* & R. K. Monson‡

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Artificial shading was provided to a Sonoran Desert ecosystem with an array of 12 regularly spaced, opaque structures. Shading resulted in a cooler, moister microhabitat below and behind each structure. Open gaps between structures also exhibited moister soils relative to a control.

Ephemeral plants increased in species diversity and showed a shift in species composition in shaded microsites, but exhibited decreased total biomass relative to controls. A deciduous shrub, *Ambrosia deltoidea*, had more mesophytic leaves, higher leaf area, carbon dioxide assimilation and growth in shaded microsites. An evergreen shrub, *Larrea tridentata*, had highest carbon dioxide assimilation and growth in sunny microsites within the array of shading structures. The plant responses observed in this study illustrate the contrasting adaptations of different desert life forms from the same habitat.

Introduction

Low water availability is considered the most important environmental factor controlling biological processes in arid and semi-arid ecosystems (Noy-Meir, 1973). Arid ecosystems differ markedly from humid ecosystems when examining limiting factors, as light availability is usually the primary environmental factor controlling primary production and concurrent energy flow to other trophic levels in humid ecosystems. Many studies have examined the effects of shading on photosynthesis, growth, morphology and phenology of native plants (e.g. Boardman, 1977). However, shading studies in arid ecosystems are few, because light is assumed not to be a limiting factor; rather, arid zone studies have focused upon adaptation to and/or avoidance of high light and, hence, high-temperature environments (e.g. Hadley, 1972).

Shading studies in desert ecosystems have addressed localized impacts beneath small trees and shrubs and have shown that shaded micro-environments exhibit denser populations of herbaceous plants than adjacent interspaces (Muller, 1953; Patten, 1978). These micro-environments are generally characterized by greater water infiltration rates (Pressland, 1976), reduced soil temperature and higher soil moisture content during dry seasons (Shreve, 1931; Patten, 1978). Shaded soils exhibit higher organic matter and nutrient contents (Garcia-Moya & McKell, 1970; Nishita & Haug, 1973), and frequently have a

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surface litter layer, which modifies soil heat flux and reduces surface evaporation. Thus, numerous interacting differences between shaded microsites and open interspaces tend to confuse interpretation of specific driving variables related to plant abundance beneath arid zone trees and shrubs.

Because of a lack of controlled experimental analyses of shaded microhabitats in desert ecosystems, this study was conducted to determine the responses of an arid ecosystem to large-scale shading of the ground surface. Artificial shading was imposed by a regular array of large, opaque panels designed to simulate a solar collector array by approximating the shape, tilt and spacing projected for collectors or mirrors in future solar power facilities. Thus, the experimental site was also utilized to provide preliminary information on the potential microhabitats that could result from the presence of solar collector arrays anticipated to cover large tracts of desert land in the next several decades (Smith, 1981).

The overall goal of this study was to analyze the effects of shade and wind deflection in a desert ecosystem, with special reference to the commonly observed increases in plant density and productivity in naturally shaded desert microhabitats. Presented here is a summary of the near-surface climatic changes and vegetational responses that occurred in a Sonoran Desert ecosystem as a result of shading by an array of structures. Microclimatic parameters examined include primary response of incident short-wave radiation and secondary responses of net radiation, environmental temperatures and soil water potential. Vegetational parameters examined include density, diversity and biomass of winter–spring ephemerals, and morphology, gas exchange and above-ground productivity of the perennial shrubs on the site.

Methods

Study site and array description

The study site was located in a level desert area several kilometers south of Tempe, Arizona (33°21′N, 111°57′W). Perennial vegetation is composed almost entirely of *Ambrosia deltoidea* (Torr.) Payne (Compositae) and *Larrea tridentata* (DC) Coville (Zygophyllaceae), with *Ambrosia* being dominant. The site is a level floodplain near an ephemeral wash, with sandy to sandy–loam soils.

The array of panels consisted of 12 iron frame structures, with wood inset, each 3·7 m wide (vertical) and 4·9 m long (horizontal), tilted 45° from vertical and capable of being manually rotated about a vertical axis. The panels were arranged in an array with the central structures facing due south and the peripheral structures oriented in east- and west-facing directions in order to cast long shadows into the site during morning and evening hours. Each panel was painted white on both the upper and lower surfaces. For a more detailed description of the panel array, see Smith (1981).

Two distinct areas, termed ‘shade’ and ‘gap’ sites, were delimited within the experimental array, based on the amount of incident radiation reaching the ground surface. The shade sites included areas below and immediately north of each central south-facing panel, which were shaded from direct-beam solar radiation throughout the midday period. Direct sunlight contacted the surface of the shade sites only in morning and/or afternoon hours, if at all. The gap sites included areas within the array that received direct sunlight during most of the midday period but were shaded in the morning and afternoon. These two sites were compared to an open desert control area on the same floodplain. In addition to shading differences, the two sites within the array were influenced by wind deflection by the panels, whereas the control was not.

Microclimate

Radiation balance and vertical temperature gradients were monitored continuously over 24-hour periods commencing at 00:00 hours on the winter solstice, spring equinox and summer solstice. On all three dates, conditions were clear for the 24 hours. Sampling was conducted at six instrumentation masts, four in the panel array and two in the control. In the shade sites, one mast was placed directly beneath a panel, while a second mast was placed just north of the panel so that it was predominantly exposed to open sky. Control masts were placed at a distance adequate to eliminate microclimatic influences from the array. All masts and associated instrumentation were placed in open areas between shrubs.

Solar irradiance was measured at each mast with a silicon-cell photovoltaic solarimeter placed 25 cm above the surface. An Eppley pyranometer was placed near a silicon-cell solarimeter in a clear area on each cardinal date to provide cosine-response correction factors at low solar elevations (*cf.* Selcuk & Yellott, 1962). Net all-wave radiation was measured with miniature net radiometers secured to each mast at 25 cm facing south.

Air temperatures were measured using 38-gauge (0·1 mm diameter), butt-welded, copper–constantan thermocouples secured to each mast at 10, 30 and 100 cm. Surface temperatures were monitored with 24-gauge (0·5 mm diameter) thermocouples covered by a thin layer of soil. All thermocouples were positioned about 30 cm from the mast. Soil temperatures were measured at 10 and 40 cm using thermocouple psychrometers placed in a systematic grid within the array and at random stations in the control area. All soil temperature stations were located in open areas between shrubs.

All instrumentation was connected to a multichannel data acquisition system via a permanent network of extension wire placed in PVC tubing beneath the soil surface. The data acquisition system was programmed to scan and record all instrumentation at 10-minute intervals throughout each 24-hour analysis period.

Air humidity near the ground surface was recorded on a continuous basis throughout the year using recording hygrothermographs housed in standard meteorological shelters in each site. Precipitation was recorded in the experimental array with three rain gauges, systematically placed from beneath a panel to the open area between two panels. Two gauges were also placed in the control area at opposite ends of the array.

Soil water potential was measured at 10 and 40 cm using the grid of soil thermocouple psychrometers. Readings were taken near dawn once a week throughout the year with a dewpoint microvoltmeter. All psychrometers used in the study were calibrated for 3 hours in a polyethylene glycol solution (mol. wt. 6,000) with an osmotic potential of $-2{\cdot}0$ MPa at 22°C, followed by soaking in distilled water for 12 hours, prior to placement in the field.

Vegetation

Winter–spring annual forbs and grasses were surveyed on permanent plots identified by numbered stakes. Fifteen stakes were randomly placed within both gap and shade sites, and twenty were randomly placed along two parallel lines in the control area. Stakes were placed exclusively in open areas between shrubs, to reduce shrub canopy effects on annual plant biomass and species composition (e.g. Halvorson & Patten, 1975; Patten, 1978).

Plant surveys were conducted six times during a single growing season, at post germination (January 30), early flower bud formation (February 28), anthesis (March 15), peak flowering and early fruit set (March 30), peak fruit set (April 15) and senescence (May 15). Phenology on these dates was based on the dominants in the control area. At each survey date, the density of rooted plants was determined by species in a 0·04–m^2 quadrat placed next to each stake in a specific direction. This same quadrat location was used on all six survey dates. Diversity was calculated from density data using the Shannon–Weaver H' formula (Peet, 1974). A second 0·04–m^2 quadrat, also located in a predetermined location next to each stake, was used to harvest total above-ground annual plant material on each sampling date. The harvested plant material was separated by species in the field, oven-dried at 80°C for 48 hours, separated into vegetative and reproductive components, and weighed to ±1·0 mg.

Carbon dioxide assimilation and stomatal conductance were monitored for each shrub species on consecutive clear days in March, May and July, representing contrasting seasonal conditions of irradiance, temperature and moisture stress. On each analysis day, six shrubs were sampled (two in each site). Four leaves from the outer canopy of each shrub from the control and gap sites were measured every 2 hours during the day. Six leaves were similarly sampled from each of two shrubs in the shade sites, one shrub in continuous shade throughout the day while the other shrub received periods of morning and afternoon direct sunlight.

Gross photosynthesis and stomatal conductance to water vapor were measured simultaneously with a dual-isotope porometer ($^{14}CO_2/^3HH$; Johnson, Rowlands *et al.*, 1979). Individual leaves were placed in a closed cuvette and exposed to a constant flow of radioactive gas which passed over a 16-mm^2 area of both leaf surfaces for 30 seconds. The leaf was then placed immediately in a vial containing 1 ml of 80% methanol stored on dry ice. Incorporated radioactivity was determined by standard procedures using a liquid scintillation spectrophotometer.

Stem water potential of each shrub species sampled for gas exchange, was measured at dawn utilizing a Scholander-type pressure chamber apparatus. Four small terminal shoots were randomly selected from each shrub to be sampled for gas exchange.

Seasonal patterns in new shoot morphology of *Ambrosia deltoidea* and *Larrea tridentata* were analyzed by randomly harvesting 20 terminal shoots of about 25 cm length from the outer canopy of two individuals from each species in each site. The length of each shoot was recorded and all side branches were also measured and summed. Total leaf area was determined for each shoot by removing all leaves and recording their area on a leaf area meter. Leaf and stem specific weights were determined by obtaining dry weights (80°C for 48 hours) and dividing by either leaf area or stem length.

Shrub production

New terminal stem production was measured for *Ambrosia deltoidea* and *Larrea tridentata* on a monthly basis throughout the year. Ten *Ambrosia* were sampled in the experimental array, six in the gap and four in the shade sites, and compared to ten sampled in the control area. Six mature *Larrea* were sampled in the array, three in each site, and compared to five sampled in the control area.

Stem elongation was measured for each species by marking growth points with wool yarn in a dimension analysis technique adapted from Burk & Dick-Peddie (1973). Six growth points on each shrub were marked at the beginning of the year near the shoot tip (below the second node in *Larrea*), and the distance from yarn to shoot tip measured to the nearest millimeter. All branch lengths above the yarn were also measured and summed. The total number of green leaves on marked shoots was recorded for *Ambrosia* in order to monitor leaf loss during the dry season. Shoot and branch lengths were used to predict shoot biomass (dry weight) via regression equations based on harvests of a shoot-length series conducted in March, June and October. Total shoot biomass (stems + leaves) in the evergreen *Larrea*, and stem biomass in *Ambrosia*, were predicted as a function of total stem length (shoot + branches). A second-order polynomial, zero-intercept model had the best fit to the data, and was thus used for *Larrea*, whereas a simple linear regression model proved satisfactory for *Ambrosia* stem biomass. For each species, a separate dimension analysis was performed in each site and used subsequently for predicting shoot biomass. In *Ambrosia*, leaf biomass was estimated from monthly leaf counts combined with average leaf weight data from the morphology samples.

Reproductive biomass was measured for each shrub species by counting all buds, flowers and fruits on each tagged shoot throughout the year. Harvests of reproductive structures provided average biomass estimates for buds, flowers and fruits of each species.

Data analysis

All data for each observation period were analyzed with a one-way analysis of variance, using the respective mean and variance values. Where significant, site effects at the $p \leq 0{\cdot}05$ level are indicated. Significant differences among sites were determined using Tukey's HSD test (Steel & Torrie, 1960) and are reported at the 0·05 or better level.

Results

Radiation and temperature

Incident short-wave radiation accumulated over the whole day was reduced in the shade sites to 20, 29 and 31% of the control value on the winter solstice, spring equinox and summer solstice, respectively (Table 1). Solar irradiance in the gap sites was variable, exhibiting accumulated values that were 65, 92 and 96% of the control values on the three cardinal dates.

Accumulated net radiation during the day, night and 24-hour period shows that the shade sites had a relatively low radiant energy input during the day, followed by reduced loss at night when compared to the other sites, with daily accumulated net radiation about zero on the winter solstice (Table 1). In contrast, the gap sites were similar to the control in exhibiting high daytime net radiation followed by low nocturnal net radiation on each cardinal date. The gap sites had a higher accumulated 24-hour net radiation than the control on both the equinox and summer solstice. Specifics of the diurnal trends in solar irradiance and diel trends in net radiation on each cardinal date were given by Smith (1981).

Maximum and minimum air, surface and soil temperatures on each cardinal date are given in Table 1. Air temperatures (100 cm) were essentially identical between sites on each date. Maximum surface temperature was substantially lower in the shade sites than in the other two sites on each date, but was higher in the gap than in the control sites on the equinox and summer solstice. Minimum surface temperatures were highest in the shade and lowest in the control sites on each date. Maximum and minimum soil temperatures (40 cm) were lowest in the shade site and highest in the control, never varying by more than 0·5°C during any 24-hour period.

Soil moisture

The 160 mm of precipitation that occurred during the study period fits the annual bimodal pattern characteristic of the Sonoran Desert of Arizona, in which winter and summer rainfall peaks are followed by spring and fall dry seasons. Peak winter precipitation at the site occurred in January and February, while most of the summer rainfall occurred in September (26·6 mm for the month). The shade sites under the panels received 28% of the yearly accumulated precipitation of the control site. The gap sites received 90% of the control precipitation.

Seasonal dynamics of soil water potential (soil ψ) at 10 and 40 cm depths are given for each site in Table 2. With the onset of winter rainfall, soil ψ attained $-0{\cdot}1$ MPa at both depths in January. By April, 10-cm soil ψ in the control area had decreased to less than $-6{\cdot}0$ MPa, the lower threshold limit for accuracy with the soil psychrometers used, compared to $-4{\cdot}5$ MPa in the gap and $-2{\cdot}15$ MPa in the shade sites. During the rest of the year, 10-cm soil ψ was below $-6{\cdot}0$ MPa at every site, except in August and September, when readings were taken soon after precipitation events. Site differences in soil ψ were much more apparent at the 40 cm depth, as a consistent monthly trend of highest to lowest soil ψ occurred for the shade, gap and control sites, respectively. Control sites reached $-6{\cdot}0$ MPa at 40 cm in June, staying below that level for the remainder of the year. The gap

Table 1. *Accumulated solar irradiance, net radiation and maximum/minimum temperatures for each site on three cardinal solar dates. Air temperatures were at 100 cm height, soil temperatures at 40 cm depth*

	Winter solstice			Spring equinox			Summer solstice		
Parameter	Control	Gap	Shade	Control	Gap	Shade	Control	Gap	Shade
Solar irradiance (MJ/m^2/day)	11·55	7·46	2·27	23·7	21·7	6·9	28·4	27·3	8·7
Net radiation (MJ/m^2/day)									
Day	5·29	3·67	1·24	11·33	12·37	3·63	11·99	12·89	4·02
Night	−1·76	−1·76	−1·07	−2·42	−2·43	−1·33	−1·99	−2·30	−1·31
Diel	3·53	1·91	0·17	8·91	9·94	2·30	10·00	10·59	2·71
Air temperature* (°C)									
Maximum	17·6a	17·6a	17·6a	26·3a	26·4a	26·4a	42·7a	42·3a	41·3b
Minimum	5·3a	5·3a	5·1a	8·3a	8·3a	8·3a	25·4a	25·6a	26·0a
Surface temperature (°C)									
Maximum	34·5a	23·6b	16·0c	43·7a	43·9a	28·5b	60·1a	63·9a	44·8b
Minimum	2·4	3·4	3·7	5·1a	6·7b	6·3b	24·5a	25·2ab	26·0b
Soil temperature (°C)									
Maximum	15·9a	14·7a	12·5b	29·8a	27·6b	24·7c	37·4a	35·0b	33·3c
Minimum	15·7a	14·6a	12·5b	29·6a	27·4b	24·6c	37·0a	34·7b	33·0c

* For temperature data at each date, values in a row followed by the same letter are not significantly different at the $p \leq 0{\cdot}05$ level.

Table 2. *Precipitation and soil water potential at 10 and 40 cm depths for each site. Water potential measurements were made at the end of each month*

	Precipitation (mm)			10-cm soil ψ (MPa)*			40-cm soil ψ (MPa)*		
Month	Control	Gap	Shade	Control	Gap	Shade	Control	Gap	Shade
January	35·2	33·7	7·0	>−1·0	>−0·1	>−1·0	—	—	—
February	49·7	44·6	9·4	>−0·1	>−0·1	>−0·1	>−0·1	>−0·1	>−0·1
March	19·1	18·4	5·2	>−0·1	>−0·1	>−0·1	−1·34a	−0·52b	−0·29c
April	7·5	5·3	1·2	<−6·0a	−4·49a	2·15b	−3·24a	−1·79b	−1·04c
May	1·8	1·5	0·1	<−6·0	<−6·0	<−6·0	−4·93a	−2·61b	−1·40c
June	0	0	0	<−6·0	<−6·0	<−6·0	−5·58a	−3·57b	−1·87c
July	0	0	0	<−6·0	<−6·0	<−6·0	<−6·0a	−4·08b	−2·28c
August	10·0	8·0	5·4	−2·88a	−2·52a	−2·38a	<−6·0a	−4·67a	−3·39b
September	26·6	21·8	12·6	−2·02a	−1·71a	−1·98a	−5·47a	−3·16b	−2·19c
October	3·8	4·4	1·3	<−6·0	<−6·0	<−6·0	−5·59a	−3·38b	−2·34c
November	3·7	3·8	1·5	<−6·0	<−6·0	<−6·0	<−6·0a	−3·81b	−2·49c
December	3·0	2·7	0·9	<−6·0	<−6·0	<−6·0	<−6·0a	−4·17a	−2·62b

* For each water potential (i.e. 10 and 40 cm depths), at each date, values within a row followed by the same letter are not significantly different at the $p \leq 0{\cdot}05$ level. For inequalities, individual data points were held at the value given (i.e. a value < −6·0 was assigned the value −6·0).

sites exhibited 40-cm soil ψ below −3·0 MPa from June to December, with a minimum mean soil ψ of −4·7 MPa. The shade sites remained above −3·0 MPa for most of the year, reaching a minimum 40-cm soil ψ of −3·4 MPa.

Ephemeral plants

Winter rains stimulated a pulse of winter–spring annuals. The dominant species were *Plantago insularis* Eastw. (Plantaginaceae) and *Schismus barbatus* L. (Thell.) (Poaceae). Above-ground green biomass of *Plantago* and *Schismus* is presented in Fig. 1. The gap and shade sites had reduced biomass through the early part of the year, relative to the control. Within the shade sites, quadrats that occurred in deeply shaded areas beneath each structure had very low plant density and biomass, whereas high biomass occurred in quadrats with midday shade but open sky exposure. A rapid reduction in green biomass occurred in the control area after March 30, with biomass senescence slightly delayed in the gap and shade sites.

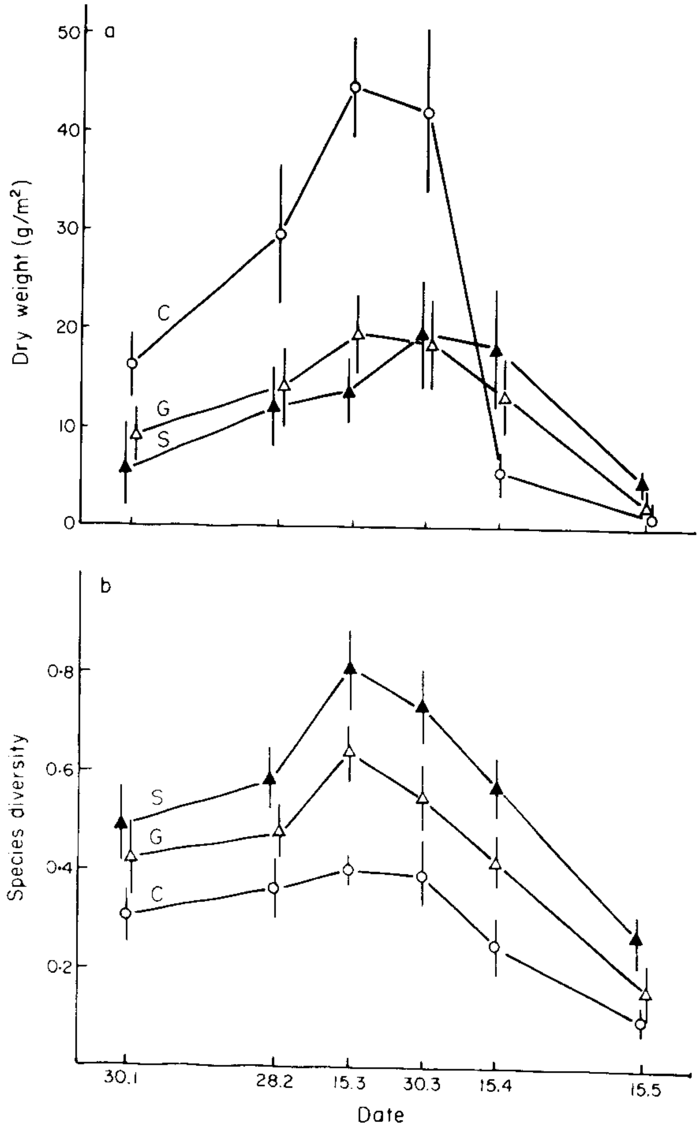

Figure 1. Seasonal changes in (*a*) above-ground biomass and (*b*) species diversity of winter–spring ephemerals in the control (C), gap (G) and shade (S) sites. Vertical bars represent 95% confidence intervals.

The ephemeral vegetation of the gap and shade sites exhibited higher species diversity than in the control throughout the growing season (Fig. 1). Low species diversity in the control area was a result of low species richness and high importance value (IV) of *Plantago* (Fig. 2). In contrast, the IVs of *Plantago* and *Schismus* were similar in the shade sites. Also, the accumulated IV of subordinate species was highest in the shade sites; subordinate species were listed and described by Smith (1981). The gap sites exhibited both intermediate species diversity (Fig. 1) and importance of *Plantago* and subordinate species (Fig. 2) relative to the control and shade sites.

Allocation of biomass to reproductive structures was seasonally delayed in both *Plantago* and *Schismus* in the shade relative to the control and gap sites (Fig. 3). Each species exhibited reduced reproductive allocation in the shade sites only in the early part of the year, eventually displaying comparable reproductive allocation at maturity. By mid-May, *Schismus* had lost most of its reproductive biomass to seed shedding in the control and gap sites but not in the shade sites.

Shrub gas exchange

Shrub water potential (ψ), relative daily stomatal conductance and carbon dioxide assimilation are given in Table 3. In March, when air and soil temperatures were moderate and soils were moist, dawn plant ψ was high in both *Ambrosia* and *Larrea*, and no site differences occurred. In the dry months of May and July, dawn plant ψ for each species was significantly higher in the gap and shade than in the control sites, where very low plant ψ's were observed. No significant differences occurred between the gap and shade sites.

Daily carbon dioxide assimilation by *Ambrosia* was highest in the control area in March, with shade plants having only 54% of that of the control plants. Assimilation by *Larrea* was comparable in the control and gap plants, but again was reduced in the shade group. By May, daily carbon dioxide assimilation was 60–70% higher in the gap than in the control sites in both species; comparable increases of 20–50% were observed in the shade sites. Similar increases were observed in gap and shade plants of both species in July. In most

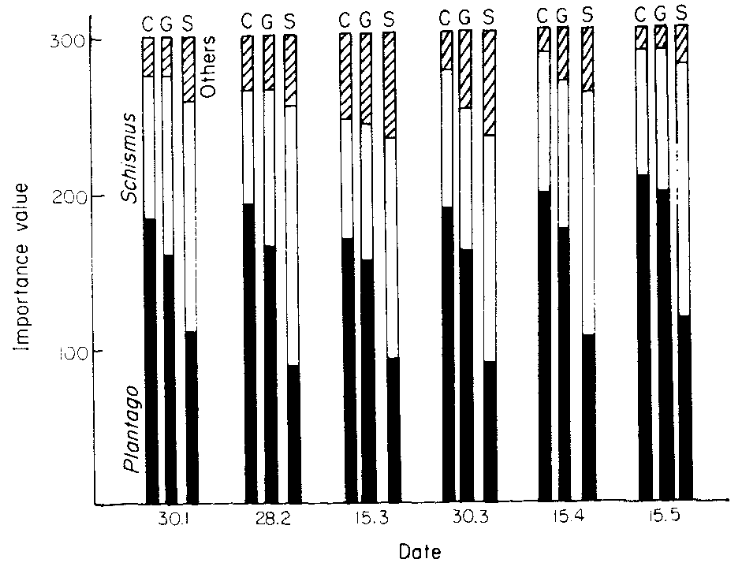

Figure 2. Importance values (IV's) of winter–spring ephemerals. At each date, the three bars designate the control (C), gap (G) and shade (S) sites from left to right. For each bar, the solid lower section designates the IV of *Plantago insularis*, the open middle section the IV of *Schismus barbatus*, and the hatched upper section the accumulated IV of all other species. IVs are the sum of relative density, relative frequency and relative biomass.

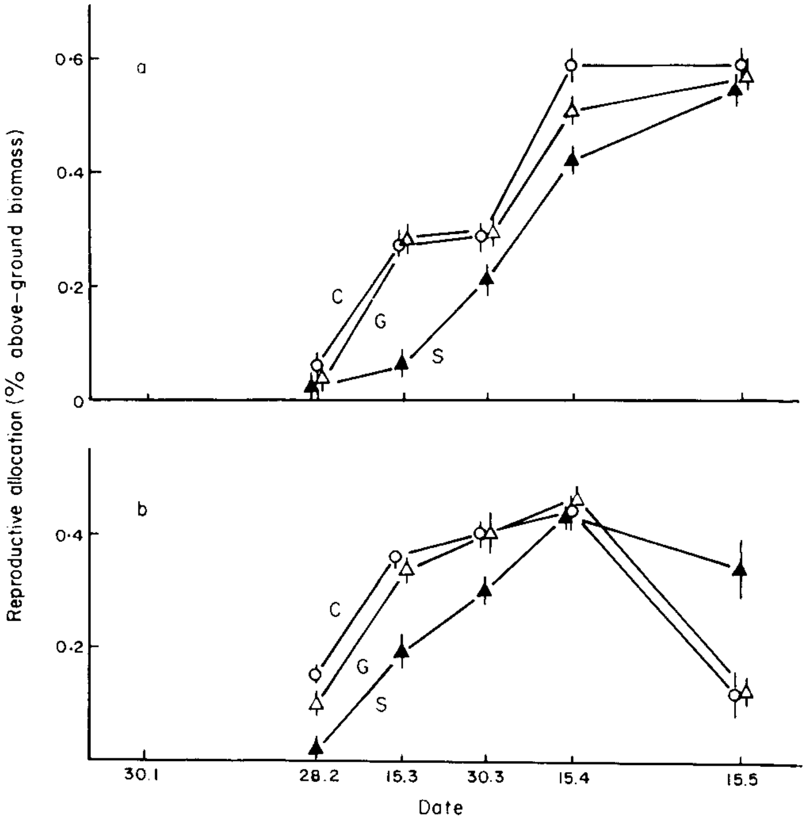

Figure 3. Reproductive allocation as a ratio of total above-ground biomass in (*a*) *Plantago insularis* and (*b*) *Schismus barbatus*. Symbols as in Fig. 1.

Table 3. *Plant water potential* (ψ), *relative daily integrated stomatal conductance and relative accumulated* CO_2 *assimilation per unit leaf area for* Ambrosia deltoidea *and* Larrea tridentata *at three times during a seasonal drying cycle*

		Plant ψ (MPa)*		Relative conductance		Relative assimilation	
Month	Site	*Ambrosia*	*Larrea*	*Ambrosia*	*Larrea*	*Ambrosia*	*Larrea*
March	Control	−0·83	−2·16	1·00	1·00	1·00	1·00
	Gap	−0·72	−1·60	0·83	1·02	0·70	1·02
	Shade	−1·31	−1·97	0·84	0·54	0·58	0·56
May	Control	−5·97a	−5·50a	1·00	1·00	1·00	1·00
	Gap	−2·85b	−3·98b	1·58	1·52	1·71	1·63
	Shade	−2·40b	−3·64b	1·58	1·75	1·23	1·48
July	Control	−6·38a	−6·51a	1·00	1·00	1·00	1·00
	Gap	−4·82b	−5·08b	1·56	1·55	1·78	1·57
	Shade	−4·19b	−4·83b	2·27	1·60	1·89	1·19

* For plant water potential in each species, at each date, values in a column followed by the same letter are not significantly different at the $p \leq 0{\cdot}05$ level.

cases, for each plant species, relative daily stomatal conductance closely paralleled daily carbon dioxide assimilation.

Shrub morphology and production

The size of individual leaves in *Ambrosia* changed only moderately with season in the control and gap sites (Fig. 4). On shade plants, a more distinct leaf-size change occurred, with the largest leaves produced during the cool, wet season (January–March). From

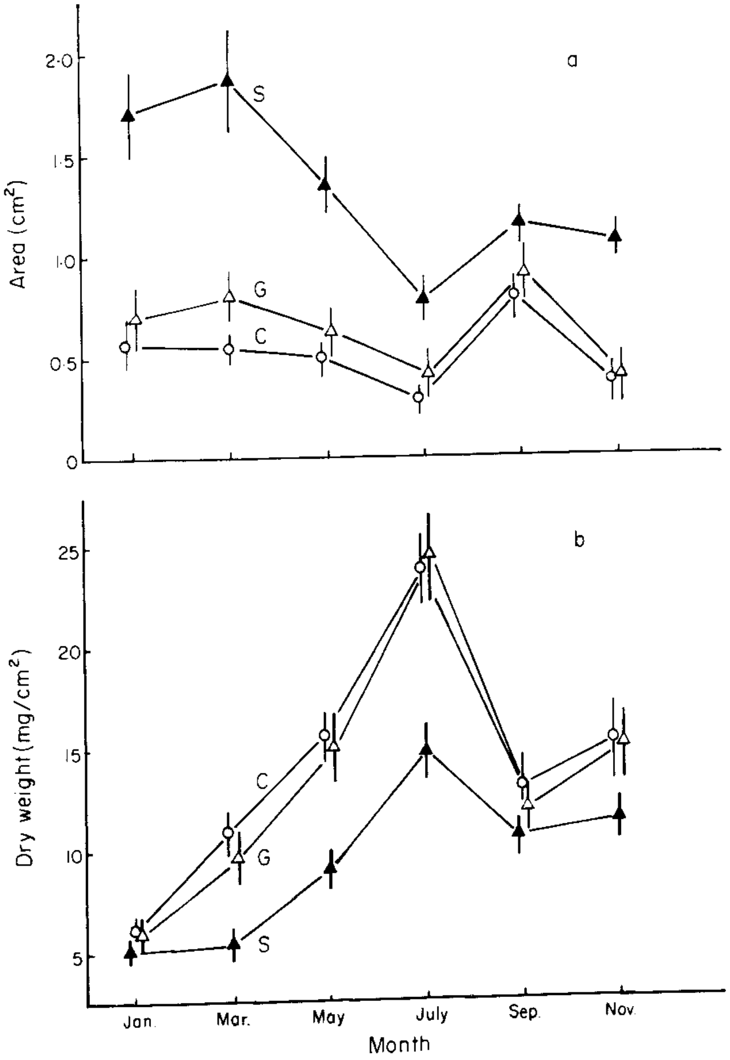

Figure 4. Seasonal dynamics in (*a*) individual leaf area and (*b*) specific leaf weight of *Ambrosia deltoidea*. Symbols as in Fig. 1.

January through May, leaf size was significantly larger on plants in the shade than in the gap or control sites ($p \leq 0{\cdot}05$). Specific leaf weight of *Ambrosia* increased into the dry season (from January to July) in the control and gap sites (Fig. 4). This increase was much less pronounced in the shade sites, resulting in lower leaf specific weights from May to July ($p \leq 0{\cdot}05$).

Significant seasonal changes in morphology were not observed for *Larrea* in any of the sites (data not shown; see Smith, 1981). Results indicated *Larrea* in the shade sites had reduced stem branching and produced stems of much lower specific weight than did controls. Shrubs from the shade sites had longer internodes and lower leaf area than did shrubs in the other two sites.

Ambrosia and *Larrea* had contrasting seasonal above-ground productivity responses to artificially imposed shade. Because of the drought-deciduous habit of *Ambrosia*, both new shoot production (stem only) and current leaf biomass (per tagged shoot) are given, along with biomass of reproductive structures, for each month of the year (Fig. 5). In March and April, accumulated stem biomass was higher in the gap than in the other two sites ($p \leq 0{\cdot}05$). From June on, *Ambrosia* in the shade sites had the highest accumulated stem biomass, with the gap shrubs intermediate. Shrubs in the control and gap sites had no new terminal stem or branch growth during the dry season (from the end of April to August), and exhibited some self-pruning during that time. Final yearly accumulated stem production was 90% higher in the shade and 53% higher in the gap sites than in the control.

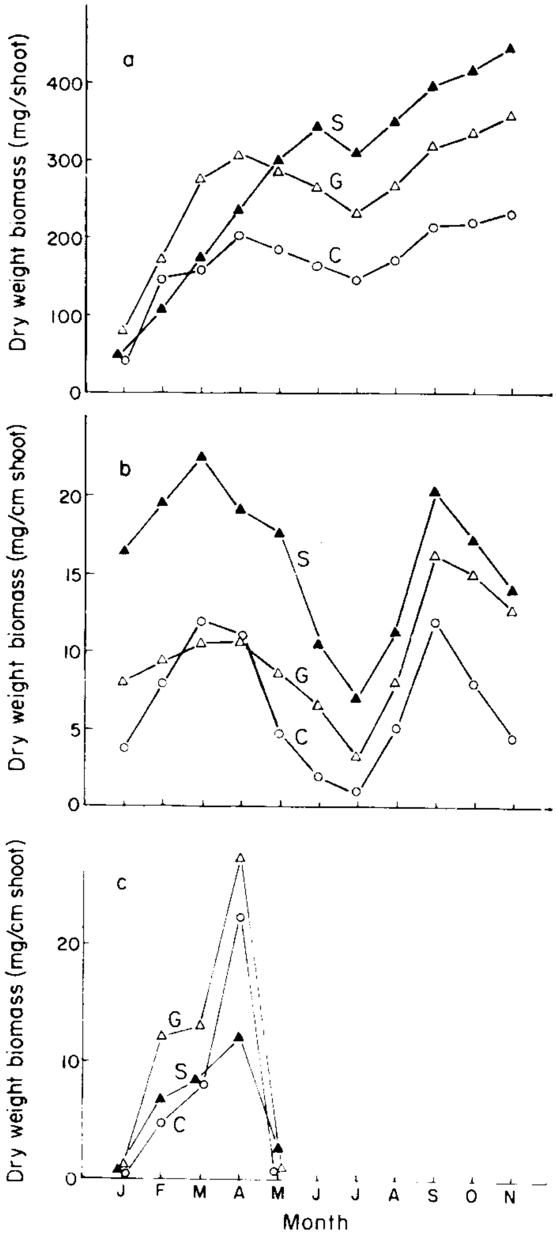

Figure 5. Seasonal changes in (*a*) accumulated stem biomass, (*b*) current leaf biomass and (*c*) current reproductive biomass in *Ambrosia deltoidea*. Symbols as in Fig. 1.

Seasonal dynamics in standing leaf biomass of *Ambrosia* followed similar trends in all study areas. Shrubs in the shade showed significantly higher leaf biomass per unit shoot length throughout the year than in the control sites ($p \leq 0{\cdot}05$ at all harvest dates). When leaf biomass was highest in the spring, the shade shrubs had almost a two-fold higher standing leaf biomass than the control and gap shrubs. By July, shrubs in the control and gap sites were almost completely leafless, while shrubs in the shade sites maintained moderate leaf biomass. When integrated over the whole year, *Ambrosia* had 23 and 14% greater leaf biomass in the shade and gap sites, respectively, than in the control.

In contrast to stem and leaf production, *Ambrosia* growing in the shade exhibited the lowest production of reproductive structures (buds, flowers and fruits) on a shoot-length basis (Fig. 5). At the April reproductive peak, 38–43% of the reproductive structures were fruits in each of the study sites, indicating that no phenological delay occurred in the shade sites. Although seed production was not quantified, total fruit production was 17% lower in the shade sites, but 59% higher in the gap compared to the control sites.

New shoots were produced throughout the year in *Larrea*, with no mid-summer depression occurring in any of the sites (Fig. 6). Accumulated new shoot biomass (stems + leaves) was highest in the gap sites from May through the rest of the year ($p \leq 0{\cdot}05$), and was 210% higher than the control value at the end of the year. Accumulated new shoot production in the shade sites was only 19% higher than in the control. Fruit production in *Larrea* was 250% higher in the gap than in the control sites. In contrast, shade shrubs produced only 20% of the fruits of control shrubs.

Discussion

Microclimate responses

The micro-environmental effects of the array of panels can be classified as direct and indirect—the direct effect of shading driving indirect responses such as temperature and soil moisture. The exclusion of short-wave solar radiation due to blockage by the structures is the most obvious and profound change that occurred within the panel array

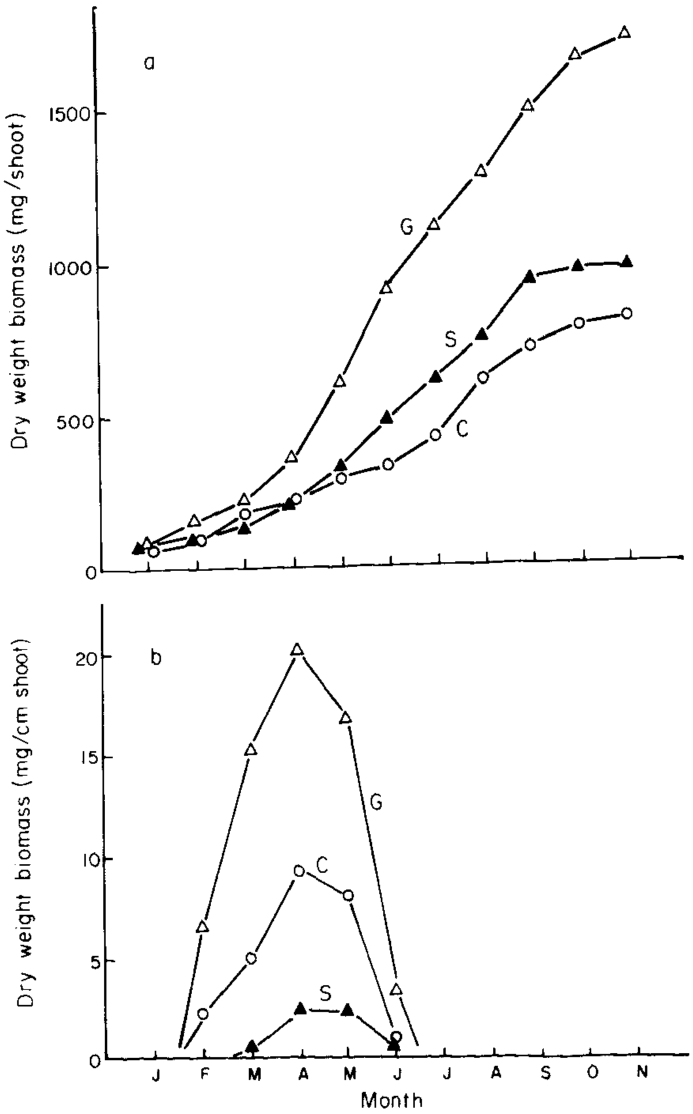

Figure 6. Seasonal changes in (*a*) accumulated new shoot (stem + leaf) biomass and (*b*) current reproductive biomass in *Larrea tridentata*. Symbols as in Fig. 1.

versus the open desert. Differences in solar irradiance between the array as a whole and the open desert, when integrated both spatially and temporally, were greatest in the winter (Table 1). On the winter solstice, all areas of the array were shaded at least half of the daylight period, and more than 50% of the array was shaded at midday. In contrast, the summer was characterized by smaller shaded areas and shorter duration of shading for specific microhabitats, but with much greater reductions of irradiance relative to the open desert (Smith, 1981). Direct-beam radiation is a more important component of total solar irradiance in the summer months (Liu & Jordan, 1960), resulting in a much larger gradient in solar irradiance between sun and shade microhabitats in the summer.

Although the shade sites had the lowest net radiation when integrated over the daylight period, they exhibited the highest net radiation at night as a result of blockage of the cold night sky by a structural surface (Table 1). As a consequence, surface maximum temperatures were lower and minimum temperatures higher than in the other two sites. Maximum and minimum soil temperatures were lowest in the shade sites throughout the year, reflecting lower radiant heating of the surface.

Soils within the panel array retained more of the soil water accumulated during the winter recharge period than did soils in the control area (Table 2). The shade sites did not follow the general trend of water depletion observed in Sonoran Desert soils, in which essentially all soil moisture available to shallow-rooted plants is lost via evapotranspiration in the dry season (Cable, 1980). From a uniformly wet profile, rapid surface drying in the spring resulted in few differences between sites in 10-cm soil ψ. Soils dried more slowly at 40 cm, where significant site differences indicated reduced losses of soil water via evapotranspiration from the shade and gap sites relative to the control. The seasonal drying rates of the three sites compare qualitatively with accumulated solar irradiance received at each site, and agree with experiments from a variety of surfaces and climates which have shown reductions in solar irradiance at the surface to result in lower soil evaporation rates (Hanks, Gardner *et al.*, 1967; Martsolf & Decker, 1970).

Ephemeral plant responses

Artificial shading in this study resulted in a lower biomass of ephemeral plants than occurred in the open desert (Fig. 2). This response contrasts with the general observation that ephemeral plant biomass tends to be highest in naturally shaded microhabitats in the Sonoran Desert (Shreve, 1931; Halvorson & Patten, 1975; Patten, 1978). However, a solid shade was produced beneath the structures used in this study, whereas shade is filtered beneath shrubs and trees. Reduced productivity in the shade sites may have been a result of inadequate photon flux densities for optimum photosynthesis, as many desert ephemerals do not light-saturate until near full sunlight (Werk, Ehleringer *et al.*, 1983). Reduced biomass in the shade may also be a result of reduction of ephemerals that normally grow in the open.

Ephemerals in the shade sites exhibited delayed phenological stages (Fig. 3) and maintained green biomass longer into the dry season (Fig. 1), apparently as a consequence of the cooler, moister microclimate. Earlier population senescence in the exposed control and gap sites was probably a water stress phenomenon (Klikoff, 1966; Beatley, 1967), occurring during the period (March–April) when 10-cm soil ψ fell from near field capacity to less than $-6{\cdot}0$ MPa (Table 2). However, 10-cm soil ψ in the shade sites fell to only $-2{\cdot}1$ MPa by mid-April, apparently causing little or no senescence of the ephemerals. This delayed senescence and phenology have also been observed in annuals from microhabitats beneath Sonoran Desert trees versus open desert interspaces (Wallace & Szarek, 1981).

Higher species diversity in the shade and gap sites relative to the control (Fig. 1) was a result of both higher species richness and a greater evenness of the two dominants, *Plantago insularis* and *Schismus barbatus* (Fig. 2). *Schismus* reaches its greatest dominance in the Sonoran Desert in association with shrub canopies in intermediate-light microhabitats (Patten, 1978). Irrigation also appears to enhance the importance of *Schismus*, relative to *Plantago*, in an ephemeral flora (Szarek, Smith *et al.*, 1982). Thus, both lower light and higher soil moisture may have resulted in a greater importance of *Schismus* in the shade sites. The observation of higher species diversity in the shade sites agrees with other arid zone studies, which have shown vascular plant species diversity to increase as environmental stress decreases (Danin, 1976).

Annuals in the cooler, moister shade sites exhibited delayed early-season reproductive development, with comparable allocation to reproductive structures attained only after control plants had entered the senescence phase (Fig. 3). Controlled watering experiments have shown that different levels of accumulated water stress can result in variable reproductive allocation in *Plantago* and *Schismus* (Szarek, Smith *et al.*, 1982). In contrast, Bell, Hiatt *et al.* (1979) found no differences in reproductive allocation resulting from microhabitat differences within individual Mojave Desert species, while Clark & Burk (1980) proposed that timing of reproduction in *Plantago* is not greatly influenced by microenvironmental conditions. However, artificial shade delayed normal phenology of both *Plantago* and *Schismus* in this study.

Shrub responses

Ambrosia deltoidea and *Larrea tridentata* exhibited contrasting responses to artificially imposed shade that were consistent with their respective morphological adaptations for survival in the desert environment. *Ambrosia* is a broadleaf and drought-deciduous shrub, while *Larrea* is a xerophytic microphyllous evergreen shrub. *Ambrosia* begins to lose leaves when plant ψ reaches about -5 MPa (Szarek & Woodhouse, 1977), retaining a leaf canopy throughout the summer if soil moisture is available. In the control area, it exhibited seasonal leaf polymorphic behavior similar to that observed in other drought-deciduous shrubs (Cunningham & Strain, 1969; Smith & Nobel, 1977). Shaded *Ambrosia* produced larger leaves in the winter and spring, and had a reduced seasonal increase in specific leaf weight into the dry season (Fig. 4). Larger leaves of lower specific weight are produced in response to low irradiance or high soil moisture treatments under controlled conditions in most broadleaf desert shrubs (Smith & Nobel, 1986).

Daily carbon dioxide assimilation in control shrubs was highest in both *Ambrosia* and *Larrea* in the spring, coinciding with maximum seasonal plant ψ. Reduction of daily carbon dioxide assimilation in the summer dry season was much more pronounced in *Ambrosia*. *Larrea* is very tolerant of water stress, maintaining positive net photosynthesis down to very low plant ψ (Odening, Strain *et al.*, 1974). Both shrub species showed reduced daily carbon dioxide uptake in the gap and shade sites in the moist spring (Table 3). However, after the onset of plant water stress, shrubs in the gap and shade sites had higher carbon dioxide uptake than in the control area, with highest relative rates of carbon dioxide uptake in July, at the peak of the dry season. By nature of its broadleaf, deciduous habit, *Ambrosia* may not exhibit photosynthetic light saturation until at or near full sunlight (*cf.* Smith & Nobel, 1986). Similarly, *Larrea* exhibits optimum photosynthetic performance under full sunlight conditions (Reynolds, Cunningham *et al.*, 1979). However, reductions in solar irradiance and, hence, photosynthetically active radiation, were apparently more than offset by the benefits of higher plant ψ in the dry season in shade site shrubs.

Seasonal new shoot production in *Ambrosia* correlates well with seasonal variation in carbon dioxide assimilation (Szarek & Woodhouse, 1977) and with plant ψ (Halvorson & Patten, 1974). Similar relationships occur in *Larrea* (Oechel, Strain *et al.*, 1972). New shoot productivity in *Ambrosia* was highest in the gap and shade sites where plant ψ was highest, and daily carbon dioxide uptake was highest in the dry season (Fig. 5). Maintenance of higher leaf area in *Ambrosia* in the shade sites apparently resulted in higher productivity on a per shrub basis. New shoot production in *Larrea* was highest in the gap sites (Fig. 6) where plant ψ was higher than in the control area and where more sunlight than in the shade sites resulted in higher daily carbon dioxide assimilation.

Reproductive growth was greatest in the gap and least in the shade sites for each species (Figs 5 and 6). Low reproductive allocation in the shade sites was, apparently, a response to low light. Additionally, the lack of water stress during the spring reproductive period may have favored the continuation of vegetative growth of each shrub species in the shade sites. For example, irrigation of *Larrea* during periods of active growth may result in a reduced percentage of biomass being allocated to reproductive structures (Cunningham, Syvertsen *et al.*, 1979).

Basic and applied implications

Although numerous studies have analyzed the biotic responses (particularly of herbaceous plants) in micro-environments beneath arid zone trees and shrubs, there have been few systematic accounts of associated microclimatic changes. Because the shading structures utilized in this study produced complete blockage of direct short-wave radiation, the microclimates produced here are most similar to microhabitats shaded for a majority of the day by an adjacent opaque structure, such as a cliff face or overhanging wall (e.g. Abd El Rahman & Batanouny, 1966; Nobel, 1978). Results of those studies were similar to the results reported here, regarding primarily the observations of: (i) up to a 90% reduction in solar irradiance during the midday period, accompanied by negative or near-zero net all-wave radiation flux at the surface; (ii) substantial reductions in surface temperature extremes; and (iii) increased soil moisture content in the protected microsites.

The results of this study illustrate that shading can reduce evapotranspiration by a substantial amount, primarily because of an assumed reduction in surface evaporation. However, shaded microhabitats beneath trees and shrubs, which support dense stands of herbaceous plants, would be expected to have much higher transpirational water loss than in adjacent interspaces. Indeed, observations have shown both an increase (Shreve, 1931) and a decrease (Cable, 1977) in soil moisture beneath desert trees relative to open interspaces. Although total water depletion may be similar in protected and open microhabitats, a greater percentage of total evapotranspiration can be expended via transpiration in the shaded microhabitat, and so more water should be available for plant activity in shaded sites.

The microclimatic changes that occurred in response to artificial shading were of sufficient magnitude to effect significant biotic changes. For ephemeral life forms, a distinct shift occurred in the shaded microhabitats in only 3 years, toward a more diverse flora and codominance of a forb and a grass. The similarity of the species composition of annual plants in the shade sites with the annual flora observed beneath trees and shrubs in the Sonoran Desert, indicates that species composition may be microhabitat-specific. More importantly, the results of this study provide substantial insight into how rapidly changes in desert vegetation can occur in response to the creation of new microclimates. It appears that dynamic processes in Sonoran Desert vegetation are not rate-limited by an inherent inability to respond to sudden changes in microclimate. Rather, the rate-limiting step appears to be the presence or availability of the propagules of plant species capable of exploiting new microclimates. For example, two ruderal species with high moisture requirements (*Baccharis* sp. and *Hordeum* sp.) became well established in the shade sites after only 3 years (Smith, 1981). These successional trends illustrate how rapidly a desert plant community can respond to sudden microclimatic changes.

The results of this study illustrate that vegetative responses to artificial shading in the Sonoran Desert are not unidirectional, but rather reflect the complexities of contrasting adaptive strategies of desert plants in the natural habitat. As a result, various life forms may be expected to exploit different microsites in the environmental mosaic, or to have contrasting periods of optimum activity. Within the ephemeral life forms, heavily shaded microhabitats beneath large trees tend to support a diverse, mesophytic flora, often of high biomass. Open interspaces tend to support sparser, often monospecific ephemeral populations. Within the perennial component of the vegetation, drought-deciduous shrubs occur more commonly in the vicinity of ephemeral washes and in protected microsites. Xerophytic evergreens such as *Larrea* reach greatest importance in the vegetation on level plains, where they are exposed to substantial periods of water stress. It is thus not unexpected that the more mesophytic *Ambrosia* had a greater positive response to shade and enhanced soil moisture than did *Larrea*.

At the applied level, these results indicate that a considerably more moderate microenvironment will occur in the vicinity of large shading structures, such as in large-scale solar collecting facilities. This may be important if native or introduced vegetation is used to stabilize soil surfaces within these facilities, as reduced surface temperatures and greater soil water conservation in the dry season would enhance the stabilization effort. The potential impact of shading by solar collectors would vary for different species. Long-term shading may not be advantageous to some shrubs adapted to the water-limited, high-light environment of the open desert, as evidenced by the negative response of the xerophytic, evergreen *Larrea* to the shade site microclimate. However, increased leaf retention by *Ambrosia* indicates that other shrubs have the potential for a substantial positive response to such shading. Therefore, the installation of large arrays of solar panels over an undisturbed desert landscape may result in significant shifts in dominance within the *Larrea–Ambrosia* community type, possibly increasing the patchiness and diversity of the plant community.

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Attachment F

	**Photograph** #1
	**Photograph** #2

Photograph #3

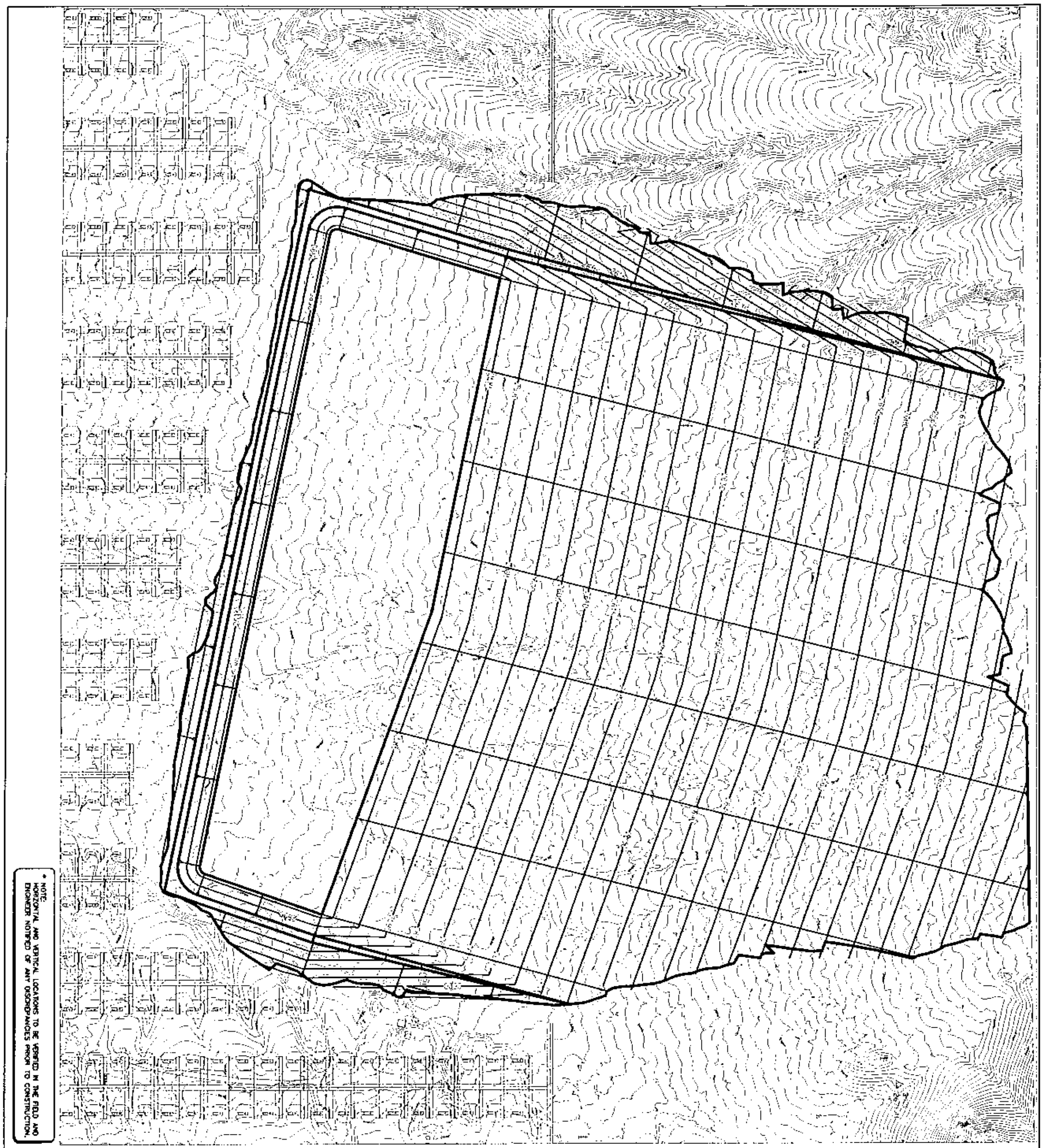
Photograph #4

Photograph #5

Photograph #6

Photograph #7

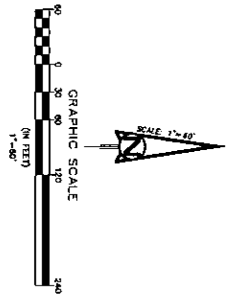
Attachment G



* NOTE:
HORIZONTAL AND VERTICAL LOCATIONS TO BE VERIFIED IN THE FIELD AND ENGINEER NOTIFIED OF ANY DISCREPANCIES PRIOR TO CONSTRUCTION.

LEGEND

▨	30' SITE ACCESS ROAD
▨	24' SITE ACCESS ROAD & ARTERIAL ROAD
▨	12' PERIMETER ROAD
■	12' MAINTENANCE ROAD
■	15' MAINTENANCE ROAD
N.A.P.	NOT A PART
——	PROPOSED FLOWLINE
	SunCatcher
— · —	EXISTING FLOWLINE NOTE: EXISTING DRAINAGE PATTERN TO REMAIN UNLESS PROPOSED FLOWLINE IS SHOWN.
	RETENTION/DEBRIS BASIN
— — —	PROPERTY LINE
— — —	SETBACK BOUNDARY
× × ×	EXISTING 100-YEAR FLOOD PLAIN
▨	LIMITS OF DISTURBANCE



NO.	DATE	REVISIONS	BY	APP.

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PHONE (909) 941-7799 • FAX (909) 941-7789

APPROVED BY:	R.C.E.	EXP.	DATE
MAURICE H. MURAD	33366	6-30-10	X-XX-09

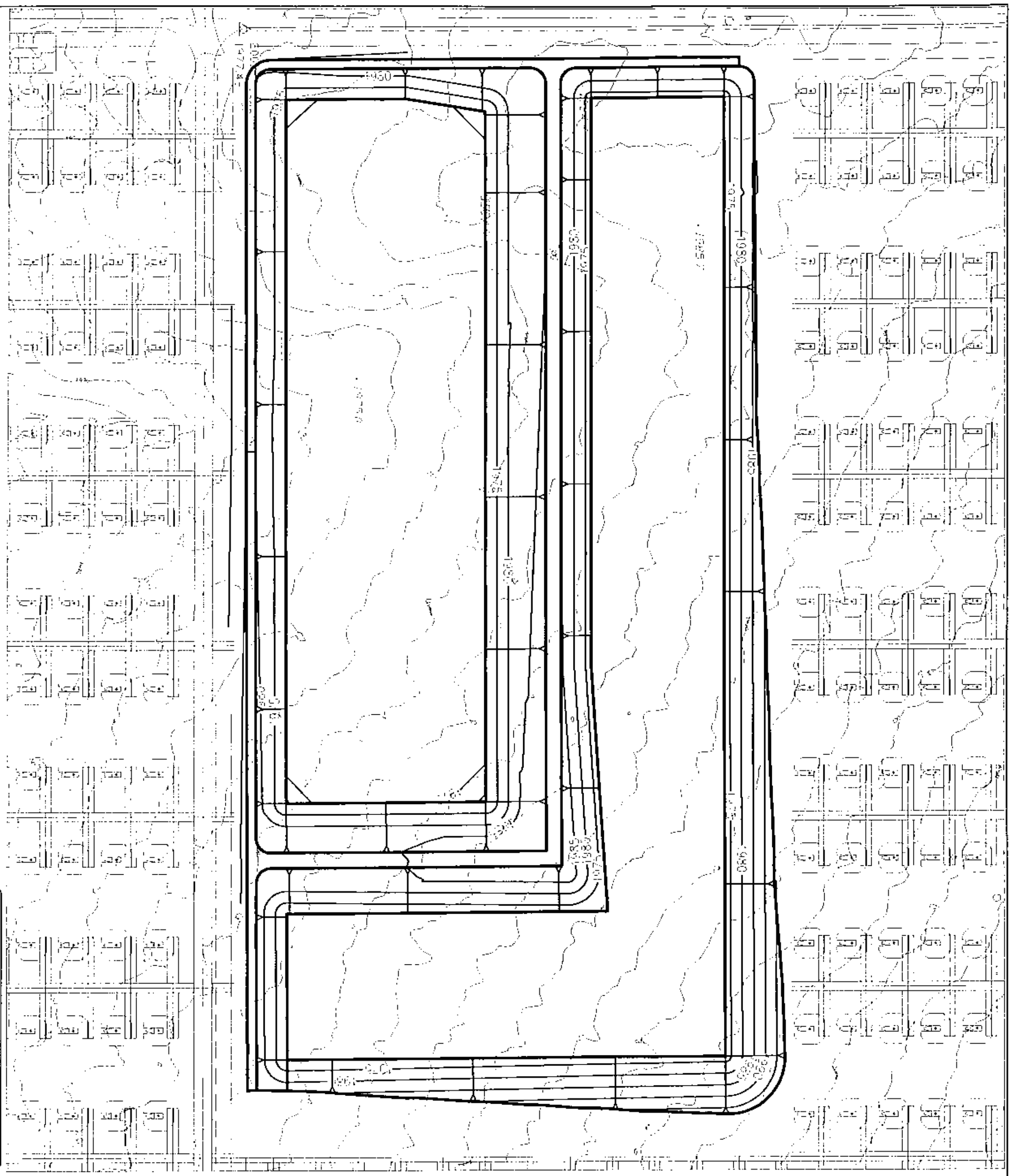
Ontario

SES
Stirling Energy Systems

TITLE:
SES SOLAR ONE LLC
SOLAR ONE – SITE PLAN
ROUGH GRADING PLAN
TYPICAL UPPER BASIN

SHEET XX OF 290

SHEET SIZE:	DESIGNED	DRAWN	CHECKED	JOB NO.	REVISION
24"x36"	M.H.M.	H-Z STAFF	M.H.M.	11034701	A



* NOTE:
HORIZONTAL AND VERTICAL LOCATIONS TO BE VERIFIED IN THE FIELD AND ENGINEER NOTIFIED OF ANY DISCREPANCIES PRIOR TO CONSTRUCTION.

LEGEND

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× × ×	EXISTING 100-YEAR FLOOD PLAIN
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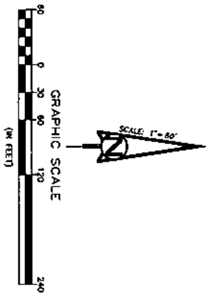

GRAPHIC SCALE
(IN FEET)
1" = 60'

NO.	DATE		BY	APP.

REVISIONS

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Ontario

SES
Stirling Energy Systems

TITLE:
SES SOLAR ONE LLC
SOLAR ONE – SITE PLAN
ROUGH GRADING PLAN
TYPICAL LOWER BASIN

SHEET XX OF 290

SHEET SIZE:	DESIGNED	DRAWN	CHECKED	JOB NO.	REVISION
24"x36"	M.H.M.	H-Z STAFF	M.H.M.	11034701	A

BEFORE THE ENERGY RESOURCES CONSERVATION AND DEVELOPMENT
COMMISSION OF THE STATE OF CALIFORNIA
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APPLICATION FOR CERTIFICATION
For the SES SOLAR ONE PROJECT

Docket No. 08-AFC-13

PROOF OF SERVICE

(Revised 12/2/09)

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*indicates change

DECLARATION OF SERVICE

I Corinne Lytle , declare that on January 29, 2009, I served and filed copies of the attached Applicant's Responses to Action Items from the 1/15/10 Workshop continuation. The original document, filed with the Docket Unit, is accompanied by a copy of the most recent Proof of Service list, located on the web page for this project at:

[www.energy.ca.gov/sitingcases/solarone].

The documents have been sent to both the other parties in this proceeding (as shown on the Proof of Service list) and to the Commission's Docket Unit, in the following manner:

(***Check all that Apply***)

FOR SERVICE TO ALL OTHER PARTIES:

sent electronically to all email addresses on the Proof of Service list;

by personal delivery or by depositing in the United States mail at with first-class postage thereon fully prepaid and addressed as provided on the Proof of Service list above to those addresses **NOT** marked "email preferred."

AND

FOR FILING WITH THE ENERGY COMMISSION:

sending an original paper copy and one electronic copy, mailed and emailed respectively, to the address below (***preferred method***);

~~***OR***~~

depositing in the mail an original and 12 paper copies, as follows:

CALIFORNIA ENERGY COMMISSION
Attn: Docket No. 08-AFC-13
1516 Ninth Street, MS-4
Sacramento, CA 95814-5512
docket@energy.state.ca.us

I declare under penalty of perjury that the foregoing is true and correct.

original signed by

Corinne Lytle

*indicates change